BIRDS AND THE EUTROPHICATION OF A SYSTEM OF SMALL LAKES

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Ph.D 2008

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Birds and the eutrophication of a system of small lakes

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Abstract

Brown Moss is a Site of Special Scientific Interest and a part of a RAMSAR site located within a small catchment area in North Shropshire. Despite its small size, it is important and has ostensibly high ecological values. It comprises a series of standing water bodies that are crucial natural habitats for rare aquatic plants and animals such as *Luronium natans* (Linnaeus) and *Triturus cristatus* (Laurenti). However, located within such small catchment area, Brown Moss may have been eutrophicated by nutrients from land use and other factors around the site and this can lead to reduction of importance and ecological values. Being waterfowl habitats, the pools may also have become affected by nutrient input from birds and aquatic macrophytes communities may have been damaged.

Pools situated within and outside the nature reserve were investigated to assess present nutrient status. Most pools around Brown Moss were hypereutrophic with high concentrations of nitrogen and phosphorus. Pools located around the edge of the site were especially more polluted than those situated in the inner area. Nutrients in pools appeared to come from various sources. Surface run off and shallow ground water contributed most of the nitrogen to the site. Phosphorus originated mostly from the sediment within the pools and was mostly released in summer. Sediment analysis and nutrient experiments also confirmed the importance of sediment. Birds in particular and rain water were also important ultimate sources of nutrients of Brown Moss.

Although the pools are geographically located in the same small area, they differ in limnology. Some pools were plant dominated while others were turbid with phytoplankton. Both regional and local factors appeared to be involved. Water depth in pools changed seasonally in the same pattern suggesting that the hydrological cycle is essentially driven by climate. Land use appeared to have a stronger effect on pools near the fringe of the site. Pools with macrophyte dominance and absence of fish had greater numbers of large cladocerans such as *Daphnia magna* (Straus) and *D. pulex* (Leydig) as well as a greater variety of macroinvertebrates. With the presence of macrophytes and large cladocerans, such pools were in a clear water state, as large

zooplankton feeders were effective in controlling phytoplankton. In contrast, a pool lacking macrophytes and having fish was turbid and governed by algae. Fish appeared to control structure of the aquatic community strongly, especially with the respect to larger cladocerans, and as a result smaller zooplankton such as rotifers and *Bosmina* were dominant but in capable of suppressing phytoplankton.

Aquatic birds played a key role in the lakes. The strongest impact occurred when they gathered in high numbers in winter. Congregations of over-wintering Canada geese (*Branta canadensis* (Linnaeus)) in 2004-5 resulted in immediate increased concentrations of ammonium nitrogen in the pool. Accumulation of organic matter from faeces also appeared to enrich sediments with nutrients released as they decomposed. Furthermore, waterfowl affected aquatic vegetation communities on which they fed. An enclosure experiment showed that plants in control areas with bird access were more damaged and suppressed by herbivorous birds. Waterfowl also caused reduction of biodiversity of plants as a result of feeding and uprooting. The presence of large migrating flocks of mallard and teal in winter 2005-6 caused severe physical damage of macrophytes across the main pool. The impact of waterfowl on water quality and plants appeared to be of considerable significance.

Paleolimnological studies showed that nutrients and biological production in the pools have increased in recent years thus suggesting that nutrients have been continuously added to Brown Moss. Increased phosphorus was also significantly correlated with numbers of sedimentary diatoms in the sediment column. Diatom analysis showed significant differences in term of numbers and variety among pools. *Frustulia rhomboides* (Ehrenb.) was dominant in a strongly acidic bog whereas *Melosira* was the main species in a nutrient-rich pool.

Brown Moss has suffered from high nutrient input and eutrophication. The important species such as *Luronium natans* may disappear forever from the site and populations of *Triturus cristatus* may be in danger and decline if the freshwater habitats remain in unfavourable condition. In the future, suitable management and scientific investigation is needed to protect and improve the quality of the site. Failure means Brown Moss could potentially lose its conservation and ecological values.

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Chapter 1 Introduction

Chapter I
Introduction

1.1 Ecological significance of lakes and ponds

Although fresh water covers only 2.5% of the earth's surface, it is one of the most precious natural resources and essential for all kinds of life on Earth. The majority of surface freshwater (69 %) is stored as glaciers and ice and 30 % as groundwater and soil moisture. The remaining freshwater, making up scarcely less than 1 % of freshwater on earth, is contained in the inland waters, comprising lakes, rivers and wetlands. These are major freshwater sources available and accessible to human beings (Ward, 2003). Lakes especially are important and play many roles in human history, consumption, agriculture, industrial sectors, economic development and ecosystems.

The world's population depends on water and requires clean and safe water for survival and well being. Historically, human beings have had a long relationship with freshwaters. Historical evidence shows that ancient human civilizations such as Mesopotamia emerged and flourished around rivers and waters many thousand years ago. Other major cities in the present world are also situated near water bodies for consumption, agriculture and transportation.

Global demands for the use of water are now geometrically increasing in line with population growth and economic development and thus man-made water bodies are required to serve world needs (L'vovich and White, (1990). Approximately 45,000 large dams and possibly 800,000 smaller dams have been built worldwide to supplement natural lakes and ponds (McCully, 1996; Hoeg, 2000; WCD, 2000).

Water is intensively used in industrial processes such as manufacturing, food processing and cooling of thermal power generation, and in agricultural development, water plays a major role in irrigation and in increasing agricultural production. Lakes and reservoirs are also primarily used as drinking water supplies and are used to generate electricity by hydropower. In addition, water bodies offer a full range of water-based activities such as fishing, canoeing and swimming and serve as

recreational places of beauty and inspiration for humans (Eaton, 1999; Cordell and Bergstrom, 1993).

In terms of ecological significance, lakes and ponds covering only a few hectares to several square kilometres have high environmental, conservation and ecological values (Biggs et al. 1999; Sondergaard et al. 2005). For instance, inland waters are a crucial part of the hydrological cycle that drives the movement of water throughout the hydrosphere and returns water to the Earth in forms of precipitation. Lakes and ponds also support a large variety of plants and animals. Unique physical, chemical and biological characteristics of lakes determine types and numbers of flora and fauna and make lakes rich in biological diversity. Gee et al. (1997) and Sondergaard et al. (2005) stated that lakes and ponds have relatively stagnant water favouring certain species of flora and fauna and the overall diversities measured per unit of area are high. In fact, nationally, about two thirds of all Britain's freshwater plants and animals can be found in permanent and temporary lakes and ponds (Williams et al. 1999).

Phytoplankton and zooplankton are key elements in lakes and ponds. In particular, phytoplankton are the foundation of open water food chains and major sources of food for higher trophic aquatic consumers such as zooplankton, invertebrates and fish. Phytoplankton also play a role in carbon sinks through photosynthesis and biogeochemical cycles (Longhurst, 1991; Falkowski et al. 2000) although the proportion of carbon delivered to sediment by freshwater phytoplankton is smaller than that of marine plankton. Zooplankton are also considered as important links between primary producers and consumers in the food web. They are a food source for other animals, especially fish and in addition, as they primarily graze on phytoplankton, they control microscopic algae and may prevent a turbid-water state dominated by phytoplankton (Gliwicz, 1975; Lampert and Taylor, 1985; Moss et al. 1997). Lakes and ponds also provide different pelagic, benthic and riparian habitats for wildlife such as amphibians and waterfowl. These include many threatened and protected freshwater species such as great crested newt (*Triturus cristatus* (Laurenti)), natterjack toad (*Epidalea calamita* (Laurenti)), marsh clubmoss (*Lycopodiella inundata* (Linnaeus)) and medicinal leech (*Hirudo medicinalis* (Linnaeus)) (Williams et al. 1999).

Aquatic macrophytes in lakes are also key components. Aquatic plants are important food sources and provide spawning and nursery grounds for aquatic insects and fish. Additionally they provide a refuge against predation for zooplankton and small invertebrates (Timms and Moss, 1984; Euliss et al. 1991; Lodge, 1991; Batzer et al. 1993). Freshwater plants also help stabilise a clear water state since they reduce phytoplankton growth by extracting nutrients from the water and releasing allelopathic substances (Jesser, 1995). In addition, macrophytes prevent resuspension of the sediment by waves and wind (Meijer et al. 1990) and protect shorelines against erosion and collapse.

Lakes also act as indicators of change. For example, substantial assemblages of phytoplankton that often change water colour can indicate excessive loss of nutrients from the land (Rawson, 1956; Prygiel and Coste, 1993; McCormick and Cairns, 1994). Aquatic invertebrates may also be used as biological indicators to indicate contamination of trace organic pollutants or heavy metals in water bodies (Lamberti and Berg, 1995; Kazanci and Girgin, 1996). Submerged aquatic plants can also act as indicators of ecological conditions such as the trophic status of water bodies (Palmer et al. 1992; Clayton and Edwards, 2006). Furthermore, sediments in lakes are important. They provide information about past environmental change since sedimentary organic matter and remains of plants and animals in lakes, for example, can be used to reconstruct the history of climate change, nutrient loading and eutrophication and to trace the response of a lake to nutrient loading (Schelske and Hodell, 1991; 1995). This information can be used to manage and protect freshwater ecosystems.

Because of the high value and significance of lakes, lakes should be protected and maintained in good and healthy conditions for their inherent beauty and a wide range of practical benefits for both humans and other organisms. However, due to population increase and intensive land use practices, lakes are very susceptible to change and are often severely polluted and misused.

1.2 Problems of small lakes/ponds in both global and United Kingdom settings

Addressed by the United Nation Environment Programme and by the Millennium Ecosystem Assessment, the world's most critical freshwater issues are negative changes in quantity and quality (UNEP, 2005; Hassan, et al. 2005). Since human population has doubled from 1960 until the present, water withdrawals have increased by a factor greater than six fold (Cohen, 2003; WMO, 1997). The increasing demands for water following population growth are due to increase in food production and inefficient agricultural practices, economic developments that change people's lifestyles and expansion of human settlements. Furthermore, rise in temperature affecting aquatic ecosystems is of concern nowadays. Climatic warming will result in greater evaporation in summer and greater winter precipitation (Hengeveld, 1990; Magnuson et al. 1997; Schindler, 1997). Thus, reduction in availability and water supply may result in competition, national and international conflicts and limitations of the use of such resource.

Rise of human population not only affects available water supplies but also impacts on the quality of water. Many lakes and reservoirs are being polluted from anthropogenic sources in many parts of the world and problems seem to be becoming more serious and widespread. As a result, the ability of ecosystems to provide clean and reliable sources of freshwater is impaired and limited. Sewage from households, industrial discharge of chemical wastes with and without treatment and fertilizer use in agriculture are major sources of world water pollution and are of great concern to users of the resource. In particular, inputs of nutrients from agricultural fertilizer have risen and have had strong effects on water bodies during the past two centuries. The global production of agricultural fertilizers increased from less than 10 million metric tonnes of N in 1950 to ca. 80 million metric tonnes in 1990, and the production is predicted by some authors to exceed 135 million metric tonnes of N by 2030 (Vitousek et al. 1997b). Large quantities of phosphate minerals are also mined and processed to create P-containing fertilisers, and these fertilisers are applied in huge amounts worldwide even to soils that already contain ample reserve of P (Smith et al. 1999).

Surplus N and P accumulating in soils can be transported from the land into surface waters and then migrate into groundwater and eventually also enter lakes and reservoirs. These excessive nutrient loadings to water bodies can degrade water quality severely and lead to eutrophication problems. Eutrophication was recognized as a pollution problem in European and North American lakes and reservoirs in the mid-20th century (Rohde, 1969) and since then it has become apparent across the world. Threats of water quality degradation are most severe in areas where water is scarce because the dilution effect is inversely related to the amount of water in circulation (Hassan et al. 2005). Shallow lowland lakes especially are prone to problems of eutrophication from nutrient runoff, serving as integrators of the land uses that occur within their catchments and these ecosystems are far more numerous than deep lakes (Kelly and Jellyman, 2007).

Healthy, clear water, stable lakes and ponds, dominated by a diversity of aquatic plants and animals, are desirable (*Figure 1.1*). But once they have become dominated by phytoplankton as a result of excessive nutrient enrichment, lakes face a variety of adverse problems and undesirable changes in ecosystem structure and function. Perceived aesthetic value is also reduced. When dominated by phytoplankton and dense free floating algae, lakes lose biodiversity of vegetation and animals. The disappearance of submerged aquatic macrophytes is caused by the shading from planktonic algae, periphyton and/or floating plants (*Figure 1.1*) (Balls et al. 1989; Sally and Mitchell, 1994; Hilt et al. 2006).

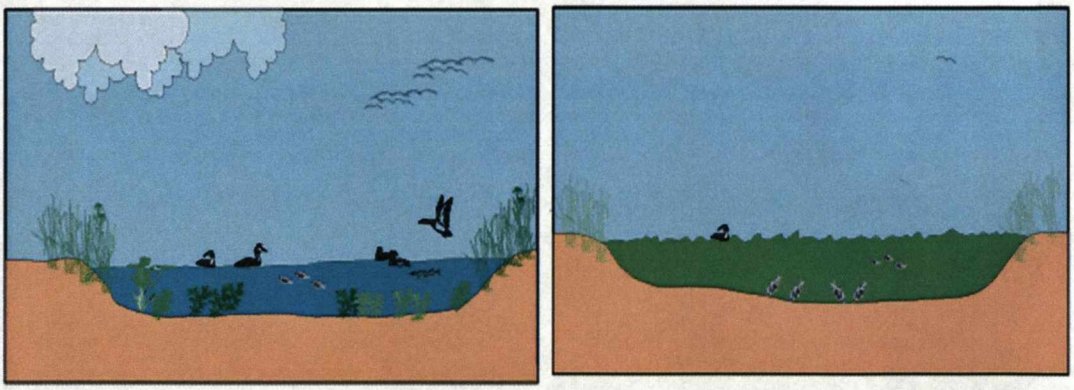


Figure 1.1 Plant-dominated clear water with high biodiversity of vegetation and animals (left) and phytoplankton-dominated turbid lake (right) with the presence of zooplanktivorous fish and absence of submerged vegetation (source : Scheffer, 1999).

Blooms of algae decrease oxygen in the water, especially in the deeper parts as a result of decomposition of dying phytoplankton that can kill fish and other animals (Brönmark and Weisner, 1992). Dissolved oxygen can also be scarce to such severe levels at night when such photoautotrophs respire and consume oxygen. Many species of animals may also disappear and may be replaced by other species, often exotics, more tolerant of the poor environmental conditions. Shorelines and littoral zones may be fouled by masses of rotting and stinking algae, causing unpleasant taste, hues and odour.

Furthermore, toxic phytoplankton, especially blue green algae, can be harmful to animals, including human beings (Anderson, 1994). There are many genera of freshwater phytoplankton such as *Anabaena, Aphanizomenon, Nodularia, Microcystis* (Paerl et al. 2001; Love et al. 2005; Kehr et al. 2006; Padilla et al. 2006) that produce toxins, especially when blooming. *Microcystis* and *Anabaena* in particular can cause skin rash and allergic symptoms among swimmers (Annadotter et al. 2004). Health-related problems in humans can also occur where eutrophic conditions interfere with drinking water treatment (Bartram and Chorus, 1999).

Impairment of aquatic resources by eutrophication can also have substantial economic effects and slow down development. Economic losses attributed to eutrophication include cost of water purification for human use, losses of fish and wildlife production, and losses of recreational amenities (Wilson and Carpenter, 1999). Restoration of lakes, damaged by eutrophication, also costs large sums of money for repair and recovery. There are tradeoffs between benefits from polluting activities and the ecosystem services that are forgone due to the consequences of eutrophication (Carpenter et al. 1997, 1999).

In the United Kingdom, the main current problems of lakes and reservoirs are similar to major issues occurring worldwide. Changes in quantity and quality of inland waters in particular are of most importance. Recently, drought problems in England have been more common, particularly in the south east which is still facing possibly the most serious drought of the past 100 years (*Figure 1.2*). The drought also causes other negative problems in lakes and small ponds in particular such as toxic algal blooms, fish kills and drying up of water bodies.

Figure 1.2 Drought at the main lake at Brown Moss, Shropshire in 2005 (left) and flood problems in the UK (right) (source: www.greenpeace.org.uk)

Many lakes and reservoirs in the United Kingdom are being polluted by sewage effluent, fertilisers and animal waste according to the report of Environment Agency (The Independent, 2007). Especially in recent years, the problems have accelerated as agricultural systems have intensified with higher stock densities and increased fertiliser use (Moss et al. 1997). Fertilisers and manure on land containing high N and P compounds can cause weeds and algae to flourish, depleting oxygen from the water at night and suffocating fish and, partly as a result, encouraging growth of alien, invasive species.

Figure 1.3 Algal bloom (left) and one of the highly eutrophic pools at Brown Moss (right)
((left photo) sources : www.cornwallriversproject.org.uk)

A recent study by the UK's Environment Agency covered the 1,047 most ecologically, valuable of Britain's 14,000 lakes and revealed that 379 of them are so degraded that they are in urgent need of rehabilitation (Environmental Agency, 2006) (*Figure 1.3*). This finding confirms Carvalho and Moss (1995) who found that 85 cases or 84% of the 102 English Sites of Special Scientific Interest showed symptoms of eutrophication. Some, such as Semerwater in the Yorkshire Dales, are severely affected and also in south east and southwest England, many shallow ponds are often eutrophic with high phosphorus concentrations (Bennion, 1994; Hendry et al. 2006).

The challenge for the twenty-first century will be to manage fresh water to balance the needs of both people and ecosystems (Hassan et al. 2005). Understanding the lakes' problems and developing techniques for reconstructing and maintaining healthy, stable lake systems are considered to be necessary for sustainable ecosystems in future generations.

1.3 General background of conservation legislation in the United Kingdom

In the United Kingdom (UK), the government has ostensibly protected and conserved species and a healthy environment through conservation legislation. In 1949, the Nature Conservancy was established by the government to identify and protect prime areas of scientific interest as representative of the remaining natural and semi-natural biological and geological areas in the country. The level of conservation ranges from individual species to areas of designated land. For example, Sites of Special Scientific Interest (SSSI) (English Nature, 2007) are apparently effective conservation units and are notified under Section 28 of the 1949 Act aiming to secure protection and appropriate management of the most important areas of wildlife habitat, to provide a resource for scientific research and to provide a resource for recreation so long as this does not compromise the wildlife habitat (Department for Environment, Food and Rural Affairs, 2008). Natural England has full responsibility for identifying and protecting the notified sites.

Natural England must notify landholders and occupiers of any land that they consider to be of special interest because of any of its flora, fauna, or geological and physiographical features. Afterwards, owners of those lands are allowed a period of four months to object to the proposed notification of a new SSSI. Any objections or issues raised by the owners within those periods are then considered by Natural England's Council. Members of the Council, appointed by the Secretary of State for Environment Food and Rural Affairs decide ultimately whether or not to confirm a notified SSSI. Once the site is notified as a SSSI, it is under legal protection.

Apart from SSSIs, local nature reserves (LNRs) and special areas of conservation (SACs) are designated to protect important sites in the United Kingdom. Local Nature Reserves are places with wildlife or geological features that are of special interest locally and statutory designation is made under Section 21 of the National Parks and Access to the Countryside Act 1949 by principal local authorities (Natural England, 2007). Special Areas of Conservation (SACs) are also strictly protected sites designated under the European Commission (EC) Habitats Directive and considered to be most in need of conservation at a European level (Joint Nature Conservation Committee, 2007).

In addition, there are also sites across the United Kingdom where freshwaters are protected by international conventions/treaties as these specific areas have high values of conservation and are important internationally. Ramsar sites, for example, are wetlands of international importance, especially as waterfowl habitats designated under the Ramsar Convention in 1971 in the Iranian city of Ramsar (Joint Nature Conservation Committee, 2007; Ramsar, 2007). The Ramsar Convention is adopted for the conservation and wise use of all wetlands through local, regional and national actions and international cooperation, as a contribution towards achieving sustainable development throughout the world.

1.4 Brown Moss

In England, there are over 4,000 Sites of Special Scientific Interest and Brown Moss is one of them. Brown Moss Nature Reserve was notified in 1953 as a SSSI and is owned and managed by Shropshire County Council (Environment Consultancy, 2001). It is regarded as a remarkable location for rare plants and animals in Britain such as water plantain (*Luronium natans* (Linnaeus)) and great crested newts (*Triturus cristatus*) that have been found there and the site was designated because of the appearance of such species.

Brown Moss, a heathland with a series of small pools, is a small area (33 ha) located to the south-east of Whitchurch, in northern Shropshire (Grid reference SJ 562395) with a small surface water catchment. It is not only a Site of Special Scientific Interest but also a Local Nature Reserve, Special Area of Conservation, and part of the Midlands Meres and Mosses Ramsar Site (Environment Agency, 2002) (*Figure 1.4*). It forms part of an internationally important series of open water and peat land sites in the North West Midlands of the UK. The characteristic habitats of the meres and mosses include open water, lakes, reed-swamp, herbaceous fen, carr woodland, wet pasture, transit mire and raised bog (Environmental Consultancy, 2001).

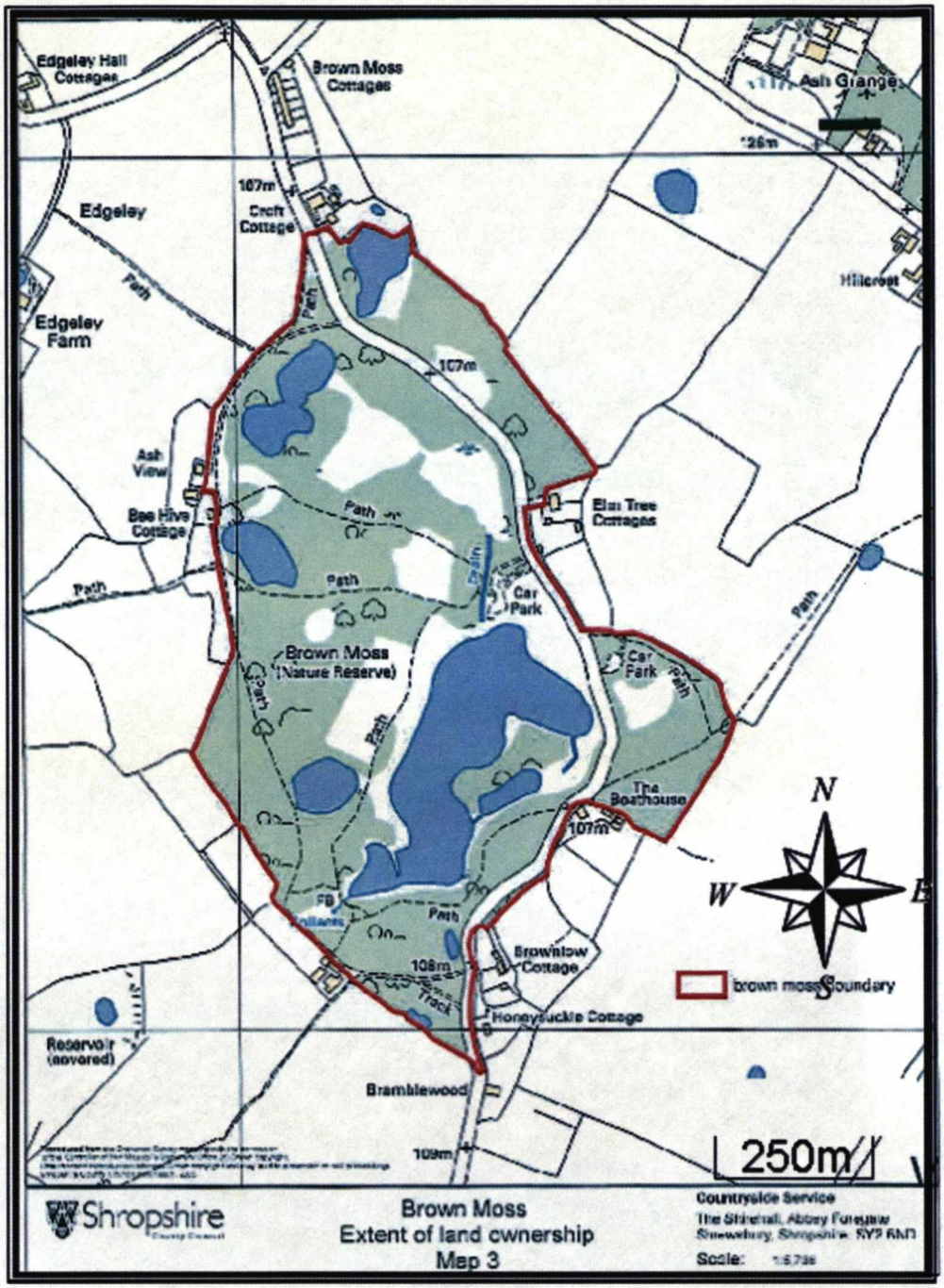


Figure 1.4 Map of Brown Moss

(Source: Shropshire County Council)

Historically, the pools at Brown Moss were probably formed by glacial activity at the end of the last Ice Age, over 10,000 years ago. Melting of icebergs can create kettle holes, good examples of which are some of the Shropshire and Cheshire meres in England (Moss, 1998). However, Brown Moss is probably not deep enough to have been formed in this way. Most likely the pools are formed in random depressions in the glacial sand deposited on the site. Sinker (1962) considered the pools in Brown Moss to be former peat bogs that had been worked out many years ago. However, there is no evidence to support this.

Figure 1.5 Aerial photograph of Brown Moss
(photo by Shropshire County)

The pools are of various sizes, shallow and occupy hollows in a substratum of sand and gravel (*Figure 1.5*). They vary considerably in their water chemistry and also in their water levels. The biggest pool known as pool 6 is located in the middle of the site and surrounded by mature woodland. It is shallow and probably charged with surface water from the immediate catchment and is perched above, and largely isolated from the deep ground water (Lockton and Whild, 2003). Most of the pools in the site depend on precipitation, sub surface percolation and a spring. Water levels of the pools fluctuate considerably (an annual amplitude of 90 cm has been recorded) and apparently independently (Environmental Consultancy, 2001) although data given in Chapters 2 and 3 suggest an opposite conclusion. Many pools have suffered very

low water levels in some years and dried up several times in recent summers (Environment Agency, 2002). The average annual rainfall for the Whitchurch area recorded between 1960 and 1992 was around 700 mm, with an average potential evaporation calculated as 575 mm (Environmental Consultancy, 2001). Because of the drying up, the fish in the biggest pool have been removed. Fish do not occur in the other pools of the SSSI either.

The land around Brown Moss is grassland and arable fields. There are crop lands and a few grazed pastures in the east, west and south of the site. There are some isolated cottages and part of the village of Ash may be included in the catchment. A small caravan park is also located in the south of the area.

The vegetation of Brown Moss is diverse. According to the Brown Moss management plan (1998-2003) and Botanical Survey of Brown Moss in 2003 (Shropshire County Council, 1997; Lockton and Whild, 2003), habitat types present on the site include woodland, carr, wetland, marginal/inundation communities, aquatic communities and heathland. Dominant and common plant species found are, for instance, pedunculate oak (*Quercus robur* (Linnaeus)), rowan (*Sorbus aucuparia* (Linnaeus)), crab apple (*Malus sylvestris* (Linnaeus)), orange foxtail (*Alopecurus aequalis* (Sobol)), great reedmace (*Typha latifolia* (Linnaeus)), soft rush (*Juncus effusus* (Linnaeus)), marsh pennywort (*Hydrocotyle vulgaris* (Linnaeus)). Brown Moss is also rich in animals. Many species are resident and other species migrate to the site seasonally. The pools support a variety of nesting and migrating water birds such as Canada geese (*Branta canadensis* (Linnaeus)), mallard (*Anas platyrhynchos* (Linnaeus)), moorhen (*Gallinula chloropus* (Linnaeus)) and coot (*Fulica atra* (Linnaeus)). Apart from birds, amphibians (*Triturus vulgaris* (Linnaeus), *Rana temporaria* (Linnaeus), *Bufo bufo* (Linnaeus)), mammals (*Apodemus flavicollis, Microtus agrestis* (Melchior), *Sorex araneus* (Linnaeus)), insects (*Coenagrion puella* (Linnaeus), *Enallagma cyathigerum* (Charpentier), *Pyrrhosoma nymphula* (Sulzer)) and aquatic molluscs (*Lymnaea glabra* (Muller)) are also found on site.

Even though Brown Moss is an important site with high ecological and conservation values, there are several issues raised by Shropshire County Council (Environmental Consultancy, 2001; Whild, 1996). In particular, there are perceived concerns over nutrient inputs from local arable/pasture fields and cottages to ponds through field drains, and sewage may discharge into the site and pools, especially following heavy rains.

Furthermore, the surrounding trees around the pools may cause problems since shading and leaf litter may influence the physical and chemical nature of the pools (*Figure 1.6*). As a result of leaf deposition and soils washing in from the surrounding areas, many pools have become shallower and this may cause ecological changes, especially in summer when pools receive less rain. Sediment accumulation may also result in the loss of open water which is a serious problem in landscape parks where retaining open water is important (Hearn et al. 2002). Grown over and unmanaged scrub on the heathland and wetland areas may lead to a loss of plant biodiversity and spreading around the pools of the non-native New Zealand Pygmyweed (*Crassula helmsii* (Berger)) may affect rare and local important marginal vegetation communities. Another issue relating to the pools is fluctuation in water levels. Because most pools receive water from rain and run off, water table fluctuations cause pools to dry up in dry summers (*Figure 1.6*) and this may cause major ecological changes, stress and disappearance of some species.

Figure 1.6 Surrounding trees around one of the pools (top, left), decrease of water level (top, right), gathering of Canada geese in the main lake (bottom, left) and public feeding of birds (bottom, right)

The site is also used for recreation. People come to Brown Moss for walking, dog walking and feeding waterfowl. Trampling by animals such as dogs and birds may damage the marginal vegetation of the pools. Feeding waterfowl may also attract more birds to the site (*Figure 1.6*) and increasing numbers of residential and exotic birds such as mallard (*Anas platyrhynchos* (Linnaeus)) and Canada geese (*Branta canadensis* (Linnaeus)) may affect communities of aquatic plants and reduce water quality.

Problems occurring at the site are of concern and its designation as a Site of Special Scientific Interest (SSSI) requires the authority to manage the site in an appropriate manner in order to maintain the interest for which it was designated (Shropshire County Council, unpublished data). Shropshire County Council is responsible for devising and instigating an appropriate management regime to ensure that decline in

ecological value of Brown Moss can be prevented and that its current value is protected and enhanced (Shropshire County Council, 1997).

1.5 Aims of thesis and issues investigated

Understanding how lakes and their physical, chemical and biological components are affected and changed is interesting and challenging because there are many factors involved. In this research I was particularly interested in the issues regarding the freshwater ecology of Brown Moss. I aimed specifically to study the pool complex with special reference to the nutrient budget in the main pool and the impact of birds that may negatively degrade water quality and change the environment of shallow lakes. Results obtained may disclose real sources of problems and can bring appropriate solutions to help improve quality of the Brown Moss. The main issues investigated in this research are described below.

1.5.1 Comparative studies of small and shallow pools at Brown Moss (Chapter 2)

Shallow pools are valuable to the ecosystem although they are small in size. They provide important food and habitats for terrestrial and freshwater species (Bratton, 1990; Whitten, 1990; Collinson et al. 1995). However, recent surveys showed that many small and shallow pools, especially in the lowlands are facing problems and loss (Boothby et al. 1995; Williams et al. 1998). One of main issues is that pools are threatened by excessive nutrients that can lead to eutrophication. Another problem regarding losses of small and shallow pools is likely due to the need of land for human settlement and agriculture or ecological succession or both (Williams et al. 1998, 1999).

In Chapter 2, I determined current water chemistry of shallow pools around Brown Moss to get an understanding of present status. In addition, comparison of seasonal changes of water chemistry between permanent and temporary pools was carried out. I also looked at the impact of variations of water levels that may cause environmental stress and affect biota and water quality of the pools. Lastly, I investigated whether there were any differences in terms of water chemistry between the pools located inside and around the edge of and outside Brown Moss.

1.5.2 Water and nutrient budget of Brown Moss (Chapter 3)

Additions of nutrients to lakes can shift the state of lakes from a clear water state to a turbid phase. Changing from clear water dominated by aquatic plants to a turbid phase dominated by algae causes serious problems (Moss et al. 1997; Padilla et al. 2006). There are many sources of nutrients that can lower quality of water in lakes. Droppings of birds contain nitrogen and phosphorus (Donald et al. 1972) and these may increase nutrient concentrations, especially when birds gather in high numbers. Ground water contaminated by sewage and rainfall-driven runoff may also bring nutrients to the lake (Brock et al. 1982; Jiao et al. 2004). In addition, nutrients from rainfall and from the sediment in the lake may be crucial and decrease quality of water and increase eutrophication problems (Allen et al. 1968; Sondergaard et al. 2003; Nowlin et al. 2005).

The aim of this study was to generate a general model for N and P load to the main pool 6. Potential sources of nutrients that may degrade water quality of the pool at Brown Moss include birds, soil and shallow groundwater, rain and sediments. I, therefore, quantified nutrient loading from waterfowl, soil and ground water, rain water and the sediment to pool 6, and investigated their seasonal impact on water quality.

1.5.3 Sediment composition and nutrient release in a shallow lake (Chapter 4)

Sediment is not only often a main net sink of nutrients but also one of the main sources (Bostrom et al. 1988; Afriso, et al. 1988; Sondergaard et al. 1993; Lynette et al. 2004; Nowlin et al. 2005). Sediments include organic matter coming from inside and outside water bodies. These include allochthonous leaf litter, soil erosion, waste excreted by animals and decaying aquatic flora and fauna (Andersson et al. 1988; Graneli and Solander, 1988; Petterson, 1998). Since nutrients can be released from the sediment to the overlying water by physical and chemical mechanisms and by decomposition of organic material by bacterial activity (Gachter et al. 1988; Sinke et al. 1990; Jensen and Andersen, 1992; Sondergaard et al. 2001), the sediment pool may be a crucial internal source of nutrients in lakes. Excessive nutrient release from sediments may thus lower water quality and stimulate the growth of phytoplankton. P

release often constitutes a major fraction of the total P load in summer (Anderson, 1982; Ryding, 1985; Jensen and Andersen, 1992) and maintains high lake phosphorus concentrations (Marsden, 1989) even when external loading is controlled and reduced.

Accordingly, I investigated composition and chemistry of the sediment from the top 15 cm layer in the main pool to gain insight into the importance of lake sediment on present nutrient storage. I also quantified key elements in the sediment that might lead to the understanding of nutrient regulation and cycling. Nutrient release from the sediment was also determined and the relationship between release of nutrients and chemical variables in the sediment was examined.

1.5.4 Limnology of Brown Moss (Chapter 5)

Regional and local factors such as climate, land use and aquatic community may have strong impact on the ecological characteristics of pools. At Brown Moss, there are several pools and water chemistry and ecological characteristics of those pools may be different. For example, different types of land use around the pools or different agricultural practices may affect water chemistry in different ways. Also, absence or presence of fish and waterfowl may be important and involved in differences of biological composition among the pools. Nutrient concentrations and primary production in pools where there are bird's habitats may be higher than those without birds (Marion et al. 1994; Manny et al. 1994; Brandvold et al. 1976). Presence of fish may cause the absence of *Daphnia* and high densities of phytoplankton, and in contrast absence of fish may result in abundance of large cladocerans, thus suppressing growth of algae (Langeland, 1977; Moss et al. 1979; Timms and Moss, 1984; Luecke et al. 1990; Burks et al. 2001).

This study aimed to find out whether the pools were mutually controlled by regional factors or locally determined by specific characteristics. Water chemistry and biological variables in particular pools were investigated to understand complexity in them. Lastly, limnological characteristics of the pools were compared and distinguished to establish what factors are involved.

1.5.5 The impact of waterfowl on aquatic plants (Chapter 6)

The biggest pool number 6 at Brown Moss is a home to resident and visiting aquatic birds. Waterfowl use lakes as their feeding and nesting grounds, especially in winter when large numbers of wildfowl migrate to lakes (Jupp and Spence, 1977; Wattaova et al. 1996; Manchester and Bullock, 2000; Arzel and Elmberg, 2004). As a result, over grazing and trampling by wild birds may cause damage to the aquatic plant community and lead to a loss of vegetation (Bolen et al. 1989; Sondergaard et al. 1996). When aquatic plants are absent, lakes lose algal inhibitors that compete with algae for nitrogen and phosphorus and this may potentially cause lakes to be dominated by phytoplankton. The problem has a wider context in that introduced bird and other species are increasingly causing problems in Europe and worldwide.

In this chapter, I investigated the impact of aquatic birds on macrophyte communities. I set up an experiment using bird cages in pool 6 to protect plants from being damaged by birds. I compared the results of the development and biodiversity of plants in cages with those in open areas where birds had access. The purpose was to quantify the damage and to compare the seasonal and long term influence of waterfowl on macrophyte communities. The results may be useful and lead to suitable management and protection of aquatic vegetation from wildfowl.

1.5.6 Paleolimnology of Brown Moss (Chapter 7)

The history of lakes can be uncovered from evidence stored in the sediments (Loffler, 1990; Williams, 1990; O'Sullivan et al. 1991; Colman et al. 1995). Physical, chemical and biological information preserved in sediment profiles can indicate the development of lakes and show how lakes have changed. Paleolimnology is widely used because past environmental data can directly be applied to problems of lake management (Smol, 1992; Hodgson et al. 1998; Reavie, 1998). For example, several studies have used organic matter in the sediment to reconstruct the trophic history of lakes (Brenner et al. 1999; Ariztegui et al. 2001) while others used remains of plants and animals as indicators of changes (Frey, 1960; Battarbee et al. 1999). Furthermore, analysis of diatoms can be applied as an effective way of inferring past pH and other conditions (Hustedt, 1939; Charles, 1984; Cook et al. 1990).

A recent study at Brown Moss showed that the site has changed. The main change detected was that eutrophic and mesotrophic aquatic plant species at the site have significantly increased (Whild, 2007). Therefore, I applied paleolimnological techniques to investigate ecological history and to determine whether there have been any changes in a trophic status of pools at Brown Moss. Basic chemistry and remains of diatoms were investigated and analysed. As there is no previous record of paleolimnology ever done at Brown Moss before, the result of this study may be of interest and beneficial for future management of the site or for further investigation.

1.5.7 Overview (Chapter 8)

This chapter summarises the over all present status of Brown Moss, the conclusions and finding of this study. Current issues at the site and suitable management are also discussed. Lastly, predicted future of the site as seen by trends in the data and future management plans for maintaining high biodiversity, stability and sustainability of limnology at Brown Moss are included.

Chapter 2 Comparative studies of small and shallow pools at Brown Moss

Chapter 2

Comparative studies of small and shallow pools at Brown Moss

2.1 Introduction

Both permanent and temporary shallow pools are important and valuable to the freshwater ecosystem because despite their small size, they can provide habitats for important assemblages of rare and endangered freshwater species as well as providing food and water to passing animals (Bratton, 1990; Whitten, 1990; Collinson et al. 1995). They are also regarded as an important part of the local character of the landscape and have an important visual and scenic value. Permanent ponds are those which contain water all year round whereas temporary ponds in contrast have an annual dry phase, usually in the order of 3-8 months, predominantly during summer and autumn (Ward, 1992; Leicestershire County Council, 1999).

The total number of ponds in the UK is not known, although a survey for the 1920s estimated that around 340,000 ponds existed in England and Wales (Wood et al. 2003). The lowest density of ponds occurred in upland areas and the highest in low areas of woodland and agricultural land in Norfolk, Suffolk and Cheshire. The Lowland Pond Survey 1996 by Williams et al. (1998), based on a stratified, random sample of 150 one-kilometre squares estimated the number of lowland ponds in Great Britain to be around 228,900 and more than one third of the ponds identified in the survey were seasonal and were dry in summer 1996. Of the remaining permanent ponds, more than 40% were very shallow, having average water depths of less than 25 cm.

Several studies have shown that small and shallow ponds are a crucial biodiversity resource (Williams et al. 1998; Oertli et al. 2002; Biggs et al. 2005). According to the Lowland Pond Survey in 1996, over half of all Britain's wetland plant species were recorded in the ponds, including uncommon species of national conservation importance and over 40% of the rarest plants recorded in the survey were found only in seasonal ponds. For wildlife, the wide variety of ponds occurring in the UK comprises an important habitat for approximately 150 of the 280 wetland invertebrates listed for conservation value in the UK Red Data Book (Duigan and

Jones, 1997). Important and rare species of aquatic flora and fauna found in ponds include great crested newt (*Triturus cristatus*), natterjack toad (*Epidalea calamita*), medicinal leech (*Hirudo medicinalis*), slender naiad (*Najas flexilis* (Wild)) and marsh clubmoss (*Lycopodiella inundata*) (Biggs et al. 2000).

Although ponds are important for aquatic organisms and the conservation and ecological value of ponds is very high, there is also evidence that many lowland shallow and small ponds are degraded or have been lost. This is mainly due to an increase in agricultural activities, expansion of urban areas and pollution that have adverse and direct effects on aquatic habitats (Leicestershire County Council, 1999; Biggs et al. 2000). Ponds have become more eutrophic due to excessive nutrient additions and as a result they have changed from healthy macrophyte dominated and clear water ponds to turbid ponds dominated by phytoplankton. Being dominated by phytoplankton and receiving high nutrient concentrations, many aquatic plants and animals are absent from such ponds and this results in loss of biodiversity and has reduced ecological value of the pond.

Furthermore, there has been a decline in numbers of shallow and small pools. Boothby et al. (1995) stated that especially in the last two decades, pond loss appears to have been greater than during any other period. Data from the Lowland Pond Survey 1996 indicated that most ponds lost between 1990 and 1996 were from arable land, while there was some increase on pastoral land (Williams et al. 1998). Another cause of aquatic habitat loss is due to natural succession, the process that occurs as a pond fills in with sediment and biological material (Williams et al. 1999). The changes associated with succession, particularly the loss of open water and the gradual reduction of water depth, have been repeated constantly throughout geological time.

At Brown Moss there is a number of permanent and temporary pools. Most of them are small and shallow and may be vulnerable to damage by human activities. Thus, the aim of this research was to determine water chemistry of temporary and permanent shallow pools inside and immediately outside the nature reserve at Brown Moss to assess current nutrient status and environmental conditions. In addition, comparison of annual changes of water chemistry between permanent and temporary

pools was carried out to assess environmental stress that may affect aquatic organisms. Lastly, I investigated whether there were any differences in terms of water chemistry between the pools located inside and around the edge of Brown Moss since the pools around the fringe of Brown Moss may be more threatened by anthropogenic impact of nearby land use than those situated in the middle of the site. The outcome of this research will be beneficial to the future management and restoration of the pools that may have suffered from excessive nutrients. A well managed and healthy pool has a greater ecological value and can support a greater variety of aquatic plants and animals.

2.2 Study pool

I chose ten permanent and temporary pools in and around Brown Moss (*Figure 2.1, 2.2*) in this study. Pools 1, 2, 3, 4, 6, 7, 10, 11 and 12 are situated within the boundary of Brown Moss, a nature reserve and a Site of Special Scientific Interest. Pool 14 in contrast is a private pool situated outside but not far from Brown Moss. Pool numbering system is based on Sinker in Brown Moss Rehabilitation Plan (Whild, 1996). Pool 5 is located next to pool 4 and pool 8 is situated near pool 7. When we surveyed the site in October 2004 we could not locate those two pools and as a result they were not included in this study.

Pool 1 is located near a cottage in the north of the site. This pool is small, shallow and in summer the pool dries up. It is surrounded by *Typha latifolia* (Linnaeus) but contained floating plants (*Lemna* spp.) at the time of taking the photograph in 2005 and no submerged plants were observed. Pool 2 is a temporary pool but deeper than pool 1. It is located by a cottage and grazed pastures. Water colour in this pool is brown. *Carex nigra* (Linnaeus) and *Sphagnum* were abundant in the southern half of the pool (Whild, 2003). Pool 3 is located in the north and opposite pool 1. This pool is shallow and also dry in summer. It is situated in a woodland and the margins of the pool have developed a species-rich fringe with *Sparganium erectum* (Linnaeus) and *Typha latifolia.* The pool also contained floating and submerged aquatic plant species.

Pool 4 is situated in the inner area of the site. The pool is shallow and temporary and dominated by aquatic macrophytes, mainly *Carex* spp, *Juncus* spp and *Polygonum*

amphibium (Linnaeus). Pool 6 is the biggest pool at Brown Moss. This pool is shallow like many other pools around Brown Moss and in some years the pool dries out. The pool is also a habitat for migrating and residential waterfowl. They gather in the pool mostly in winter. The main pool contains a variety of aquatic plants including the rare *Luronium natans,* last seen in 2006.

Pool 7 is sometimes part of the main pool 6 and shallow. During the dry season, this pool dries out and is separated from the main pool. Overgrown grey willow mostly occurs around the fringe of the pool but the pool still has open water. Pool 10 is located on the east of the site near agricultural land. A few years ago, this pool was full of overgrown scrub. In 2006 the scrub was cleared and since then the pool has more open water. Pools 11 and 12 are temporary pools. They both are small and shallow and located at the south of the site. The pools have become more over grown to the point where there is now almost no open water remaining (Whild, 2003). Pool 11 is shaded by trees which overhang its water surface and the pool is also filled by leaf litter. No submerged and floating plants have been found in pool 11. In pool 12, *Lemna* is the main plant species.

Pool 14 is a permanent pool owned by a private land owner. This pool is deep compared with other pools around Brown Moss and is a fish habitat. Submerged macrophytes are absent from the pool. Areas around the pool are local arable and pasture fields. Physical and biological variables of the study pools are presented in Table 2.1.

Table 2.1 Physical and biological variables in the study pools

Pools	Depth (cm)	Estimated area* (m^2)	Immediate land use	Cottages	Submerged plants	Fish
1	26	4,100	√	√	√	
2	29	3,600	√	√	√	
3	27	2,700			√	
4	33	3,600			√	
6	31	29,700			√	
7	26	2,900				
10	16	2,700	√		√	
11	13	220	√	√		
12	25	190	√	√		
14	54	1,200	√	√		√

* Areas of pools are estimated in relation to area of Brown Moss, which is 33 ha.

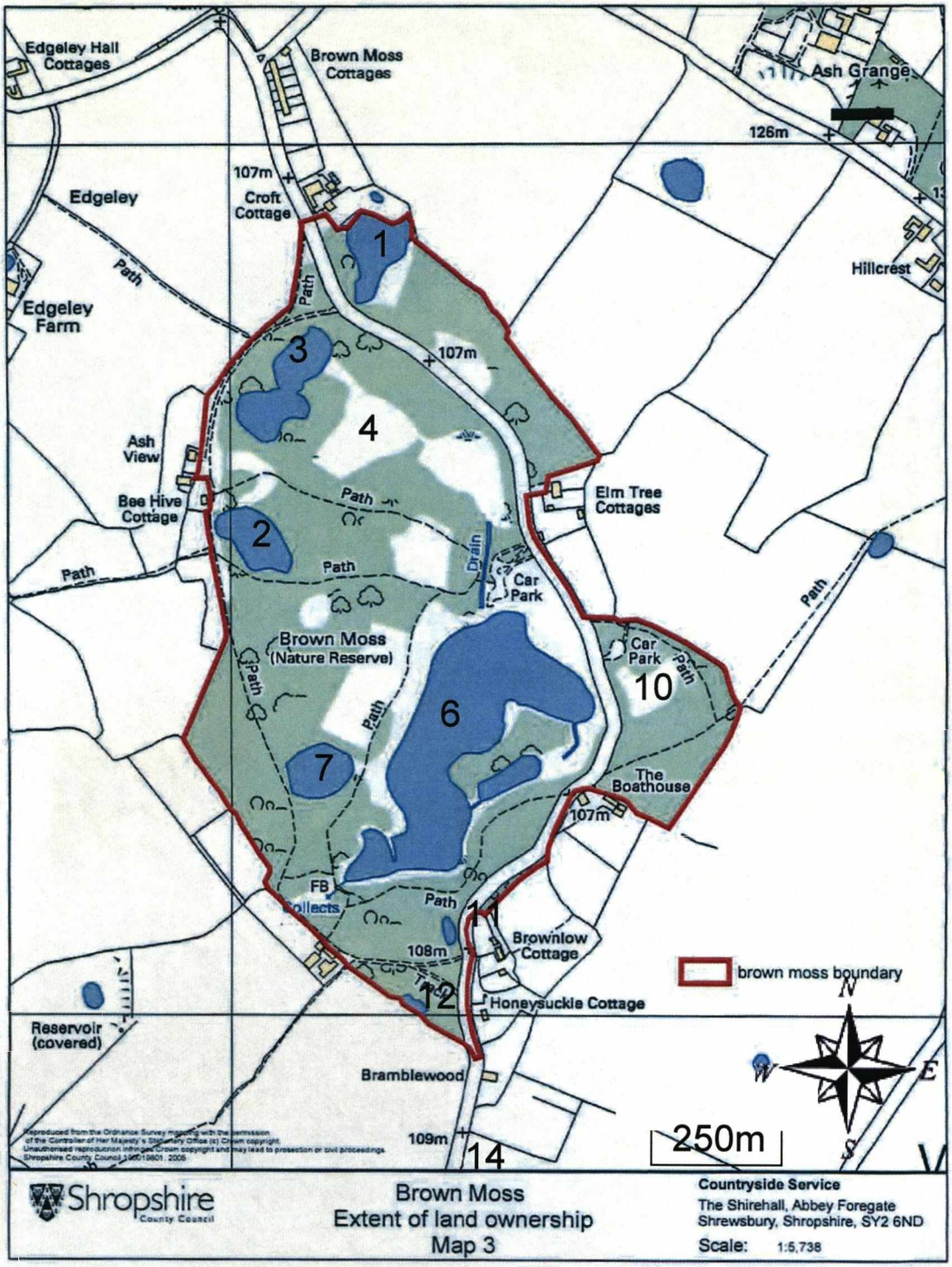


Figure 2.1 Locations of the study pools at Brown Moss

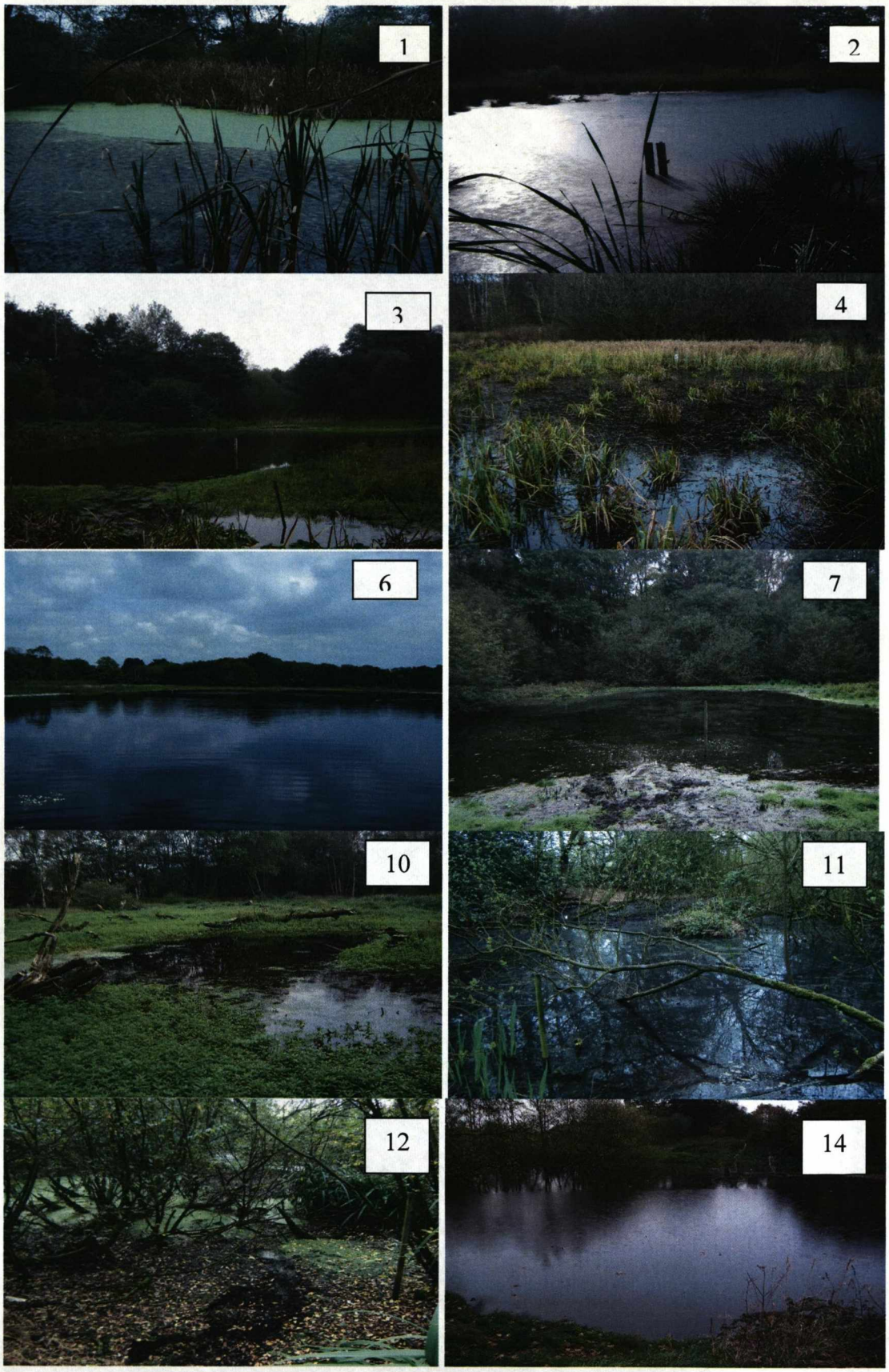


Figure 2.2 Characteristics of the study pools around Brown Moss in 2005

2.3 Methods

Water samples were collected every two weeks from the study pools from November 2004 to October 2006. Depth (cm) was measured by a ruler and temperature (^{0} Celsius) was measured by a YSI meter 550. Measurements of pH and conductivity ($\mu S\ cm^{-1}$) values using a Hanna HI 9812 pH meter were recorded in situ. Alkalinity ($mEq\ L^{-1}$) using titration against standard acid to an end point at pH 4.5 (Mackereth et al. 1989) was measured within a few hours after collection, in the laboratory. Total Suspended Solids (TSS, $mg\ L^{-1}$) were also measured by filtering water through GF/C filters and then drying the filters for 24 hours at 105 °C. Filtered water was also used for chemical analysis of soluble reactive phosphorus (SRP, $\mu g\ \ L^{-1}$), nitrate nitrogen (NO_3^--N, $mg\ L^{-1}$) and ammonium nitrogen (NH_4^+-N, $\mu g\ L^{-1}$), based on Mackereth et al. (1989). Total nitrogen (TN, $mg\ L^{-1}$) and total phosphorus (TP, $\mu g\ L^{-1}$) were measured by methods of Johnes and Heathwaite (1992).

I also measured concentrations of chlorophyll a from the study pools every 2 weeks from November 2004 – October 2006. I filtered 150 ml. of water samples through Whatman glass microfibre filters (GF/C, 47 mm) and then used a standard acetone extraction method based on the equation of Talling and Driver (1961).

I applied Tukey honestly significant difference (HSD) test for multiple comparisons among variables using SPSS 14.0 for Windows. Pearson correlation coefficients (*r*) were also used to determine correlation between TSS and chlorophyll a concentrations. Lastly, Bray-Curtis, PRIMER 6.0 was used to indicate similarity of variables among the study pools.

2.4 Results

2.4.1 Present environmental conditions and nutrient status

Average temperature of all pools at Brown Moss ranged between 8 and 19 °C and pH varied from 6 to 8 (*Figure 2.3*). Conductivity values were highest in pool 14 (468 µS cm^{-1}) and were lowest in pool 2 (110 µS cm^{-1}). Alkalinity ranged between 0.6 to 3 mEq L^{-1}. Pool 14 had highest value of alkalinity, whereas pool 2 had lowest value of alkalinity. Average TSS in the study pools varied from 0.01 to 0.05 mg L^{-1}. Pools 14 and 6 had high values of TSS whereas pools 3, 4, 10 and 11 had low TSS values.

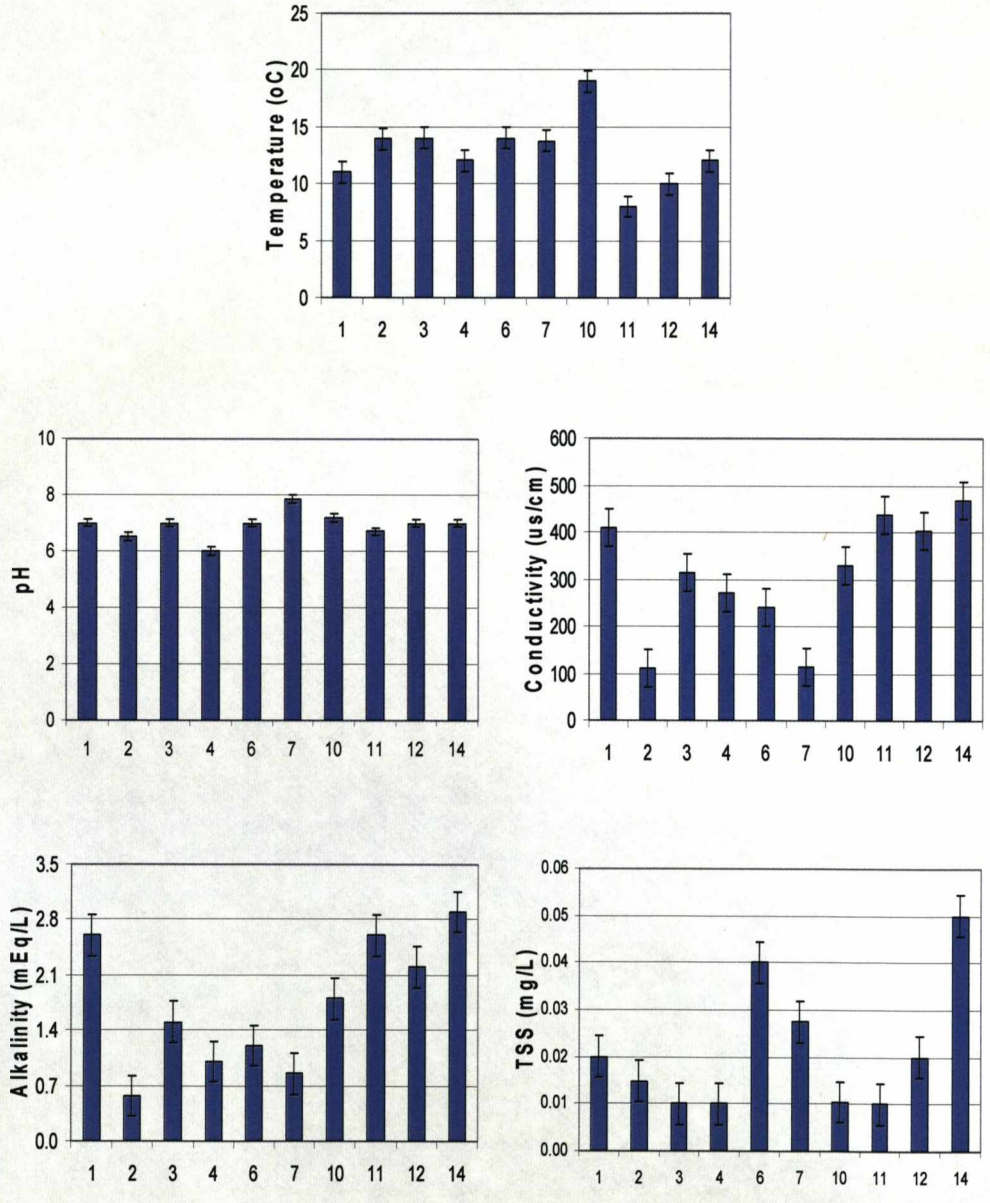


Figure 2.3 Overall means with standard errors of variables (n = 2) in the study pools

Nutrient concentrations showed that average SRP concentrations in the study pools ranged between 17 and 1,170 μg L^{-1}. The highest and lowest SRP concentrations were in pools 1 and 4, respectively (*Figure 2.4*). Pool 12 had the highest concentration of TP (1,570 μg L^{-1}) and pool 4 had the lowest TP value (102 μg L^{-1}). Mean concentrations of ammonium nitrogen in the study pools ranged from 66 to 3,700 μg L^{-1}. The highest and lowest concentrations of ammonium nitrogen were detected in pools 2 and 4, respectively. Nitrate concentrations were low in most pools except pools 11 (2.03 mg L^{-1}) and 14 (1.3 mg L^{-1}). TN values ranged from 2.4 to 6.4 mg L^{-1}. Pool 2 had highest concentration of TN and in contrast pool 3 had lowest value of TN.

Using average concentrations of TN and TP from the Florida LAKEWATCH database (Florida Lakewatch, 2000) to classify trophic state of the pools, the results showed that all pools were hypereutrophic (with TP values greater than 100 μg L^{-1} and TN values greater than 1,500 μg L^{-1}). TN : TP ratios also suggested that pools 1, 2, 7, 10 and 12 were nitrogen limited when the TN : TP ratio was less than 10. In contrast, phosphorus was limiting in pool 4 where the TN : TP ratio was greater than 17. TN : TP ratios of pools 3, 6, 11 and 14 were between 10 – 17 suggesting that phosphorus or nitrogen could be limiting. The TN : TP ratio originally comes from a study of Redfield who described the remarkable congruence between the chemistry of the deep ocean and the chemistry of living things in the surface ocean (Redfield, 1934). He mentioned that both have N:P ratios of about 16 (atoms to atoms) and when nutrients are not limiting, the molar element ratio C:N:P in most phytoplankton is 106:16:1. But in freshwater, studies of Florida lakes have shown that the ratio of TN:TP presented above indicate which nutrients play the most significant limiting roles.

Mean chlorophyll a concentrations in the study pools ranged from 14 to 140 μg L^{-1}. Most pools showed lower chlorophyll a concentrations than pools 6, 7, 10 and 14 where there were higher concentrations of chlorophyll a. The highest and lowest concentrations of chlorophyll a were in pools 10 and 4, respectively.

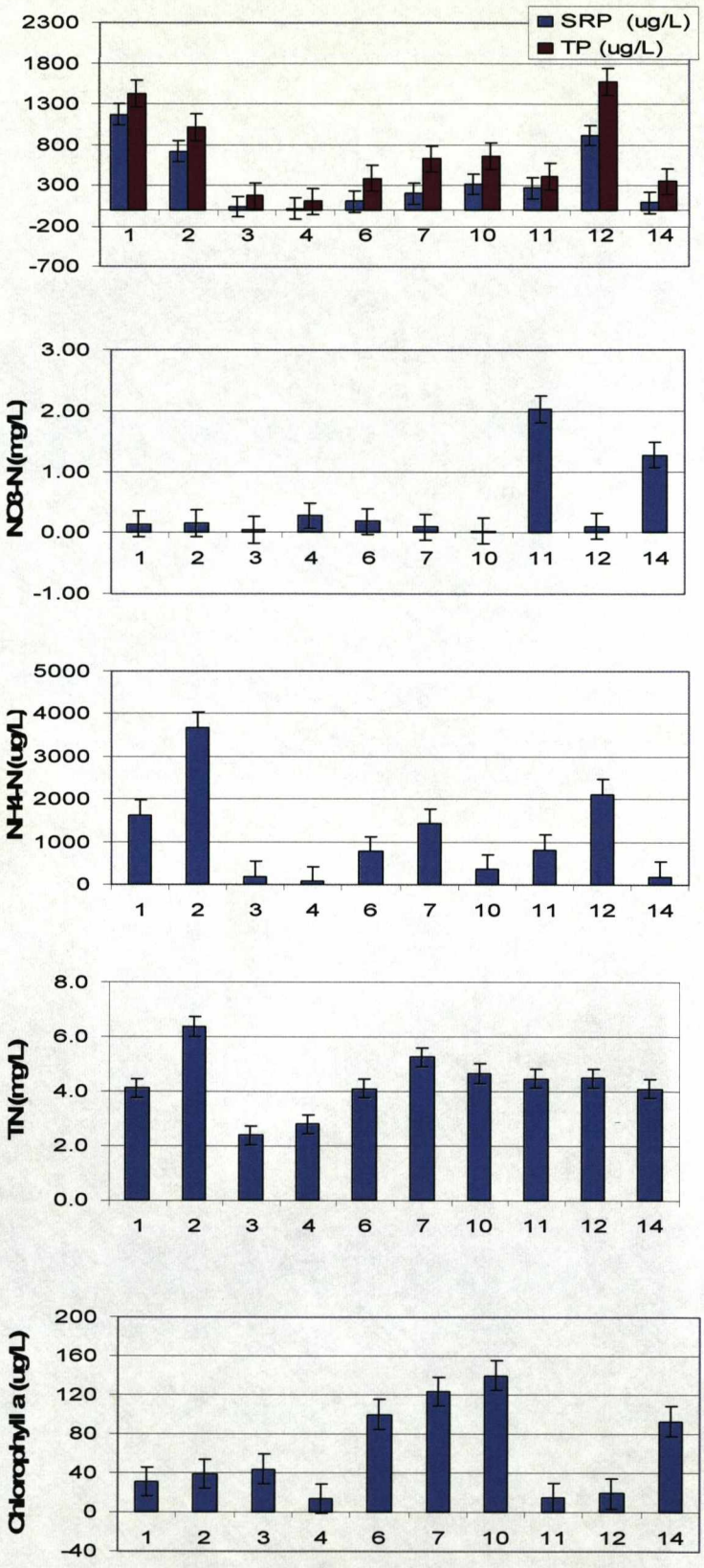


Figure 2.4 Comparative quantities of nutrients and planktonic chlorophyll a with standard errors (n = 2) in the study pools

Multiple comparisons of different chemical variables among different standing waters showed that overall SRP, TP, NO_3^--N, NH_4^+-N, TN, and chlorophyll a were significantly different among the pools (*Table 2.2*). For instance, SRP values in pool 1 were significantly different from all pools except pool 12. Statistical analysis also indicated significant differences of TP concentrations between pool 1 and pools 3, 4, 6, 7, 10, 11 and 14, and NH_4^+-N of pool 1 significantly differed from pools 2 and 4. In addition, NO_3^--N in pool 1 was significantly different from pools 11 and 14. TN in pool 1 was not significantly different from other pools and lastly chlorophyll a concentrations of pool 1 were significantly different from pools 7 and 10.

Table 2.2 Multiple comparisons of different chemical variables among pools, $*P<0.05$ and $**P<0.001$ indicate significant differences among pools (n = 49).

SRP	1	2	3	4	6	7	10	11	12
2	0.030								
3	< 0.001	< 0.001							
4	< 0.001	< 0.001	1.000						
6	< 0.001	< 0.001	1.000	1.000					
7	< 0.001	0.001	0.982	0.959	1.000				
10	< 0.001	0.462	0.887	0.839	0.978	0.999			
11	< 0.001	0.042	0.809	0.737	0.972	1.000	1.000		
12	0.347	0.856	< 0.001	< 0.001	< 0.001	< 0.001	0.440	< 0.001	
14	< 0.001	< 0.001	1.000	1.000	1.000	0.999	0.967	0.953	< 0.001

TP	1	2	3	4	6	7	10	11	12
2	0.208								
3	< 0.001	< 0.001							
4	< 0.001	< 0.001	1.000						
6	< 0.001	0.020	0.969	0.866					
7	< 0.001	0.290	0.463	0.273	0.993				
10	< 0.001	0.939	0.700	0.537	0.991	1.000			
11	< 0.001	0.071	0.972	0.887	1.000	0.999	0.996		
12	1.000	0.074	< 0.001	< 0.001	< 0.001	< 0.001	0.025	< 0.001	
14	< 0.001	0.008	0.991	0.936	1.000	0.971	0.976	1.000	< 0.001

NH_4-N	1	2	3	4	6	7	10	11	12
2	0.003								
3	0.064	0.003							
4	0.043	< 0.001	1.000						
6	0.676	< 0.001	0.969	0.918					
7	0.998	< 0.001	0.412	0.307	0.988				
10	0.678	< 0.001	1.000	1.000	1.000	0.949			
11	0.817	< 0.001	0.978	0.940	1.000	0.996	1.00		
12	0.998	0.050	0.005	0.003	0.189	0.838	0.309	0.342	
14	0.062	< 0.001	1.000	1.000	0.967	0.404	1.000	0.976	0.005

NO_3-N	1	2	3	4	6	7	10	11	12
2	1.000								
3	1.000	< 0.001							
4	1.000	< 0.001	0.998						
6	1.000	< 0.001	1.000	1.000					
7	1.000	0.001	1.000	1.000	1.000				
10	1.000	0.462	1.000	1.000	1.000	1.000			
11	< 0.001	0.042	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001		
12	0.347	0.856	1.000	1.000	1.000	1.000	1.000	< 0.001	
14	0.003	< 0.001	< 0.001	0.016	0.004	0.001	0.069	0.271	0.002

TN	1	2	3	4	6	7	10	11	12
2	0.094								
3	0.143	< 0.001							
4	0.547	< 0.001	1.000						
6	1.000	0.043	0.228	0.693					
7	1.000	0.398	0.021	0.168	0.996				
10	1.000	0.799	0.409	0.737	1.000	1.000			
11	1.000	0.262	0.159	0.532	1.000	1.000	1.000		
12	1.000	0.184	0.084	0.401	1.000	1.000	1.000	1.000	
14	1.000	0.029	0.266	0.745	1.000	0.991	1.000	1.000	1.000

Chla	1	2	3	4	6	7	10	11	12
2	1.000								
3	1.000	1.000							
4	0.999	0.985	0.940						
6	0.077	0.135	0.247	0.007					
7	0.017	0.033	0.072	0.001	1.000				
10	0.034	0.054	0.090	0.006	0.964	0.997			
11	0.999	0.992	0.964	1.000	0.018	0.004	0.010		
12	1.000	0.997	0.981	1.000	0.015	0.003	0.010	1.000	
14	0.160	0.262	0.426	0.018	1.000	0.997	0.900	0.042	0.037

Correlation coefficients between TSS and chlorophyll a among pools were determined. The results revealed that concentrations of chlorophyll a in pools 2, 7 and 11 significantly correlated with TSS values at 0.01 level. Significant correlations at 0.05 level between chlorophyll a concentrations and TSS values among pools 4 and 6 were also found. For the rest, no significant correlations between concentrations of chlorophyll a and TSS values were found.

In addition, I used alkalinity and conductivity, and nutrients (TP and TN) to investigate similarity and differences among pools (*Figure 2.5*). The result of Bray Curtis analysis of alkalinity and conductivity showed that the pools can be divided into three main groups based on values of those conservative variables. Pools 2 and 7 were in the same group that had relatively low values of both alkalinity and conductivity compared with other pools. Pools 3, 4, 6 and 10 represent medium values of alkalinity and conductivity and lastly pools 1, 11, 12 and 14 had higher values of those variables.

By using nutrients, the study pools at Brown Moss can also be divided into three groups (*Figure 2.5*). Overall, the first group (pools 3 and 4) represents low nutrient concentrations compared with other pools. Close similarities among pools 6, 7, 10, 11 and 14 were also found and these pools are in the second group since they had moderate amounts of nutrients. The last group is composed of pools 1, 2 and 12 with waters that had high quantities of nutrients.

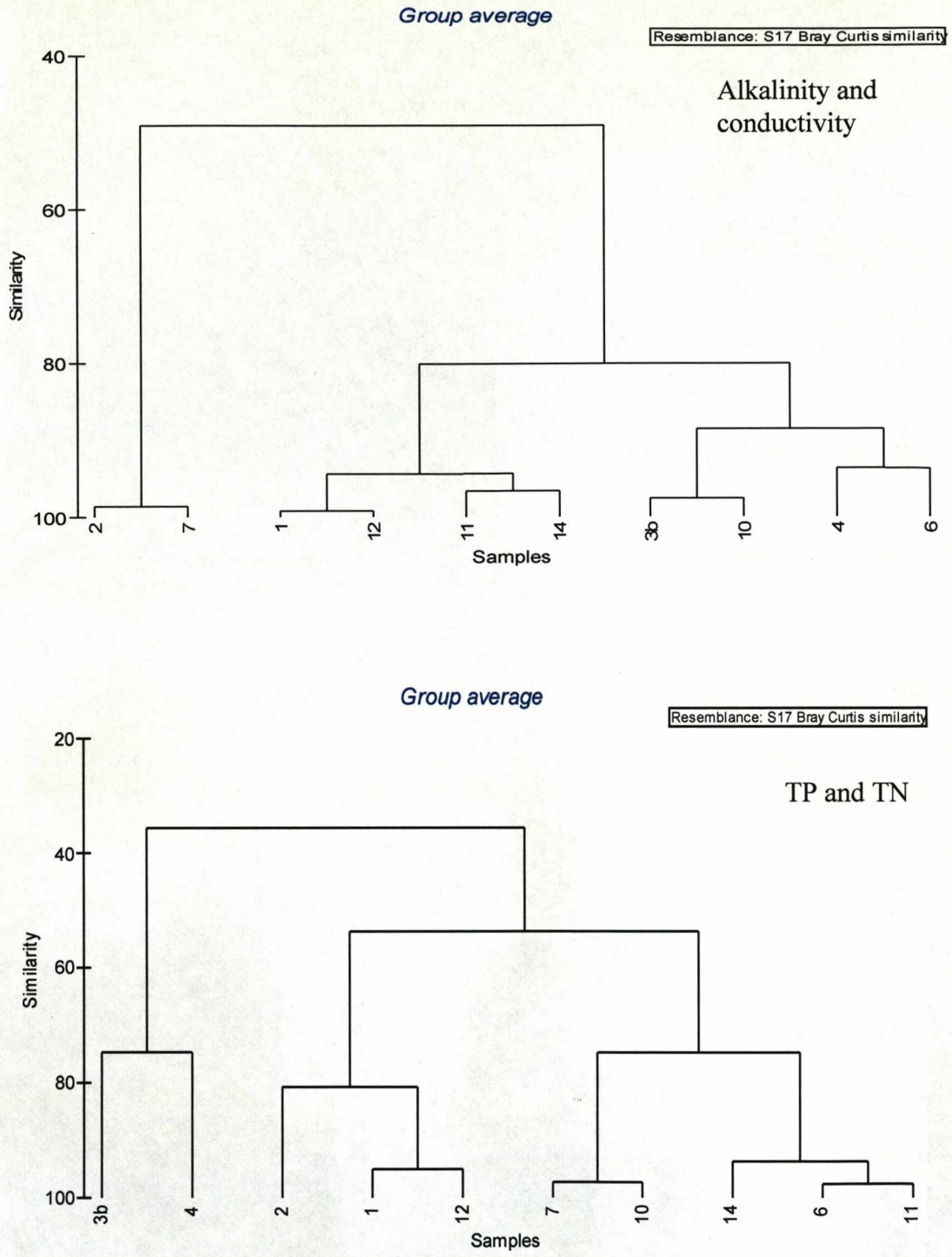


Figure 2.5 Similarities among pools at Brown Moss using alkalinity and conductivity (above) and TP and TN values (below)

2.4.2 Seasonal changes of water depth and chemistry

Most pools at Brown Moss are shallow. Especially in temporary pools, average water depth was less than 0.5 m. In the permanent pool 14, an average water depth was 53 cm and the pool contained water all year round. As shown in Table 2.3, most temporary pools dried out in summer and autumn 2005 and 2006. Pool 11 had the longest annual dry phase of up to four months.

Table 2.3 Water levels and an annual dry phase of pools in Brown Moss

Pools	Average water depth (cm)	Period of drying out
Temporary		
1	26	Sep 2005 and Oct 2006
2	29	Nearly dried out (3 cm. in Sep 2005)
3	27	Nearly dried out (6 cm. in Sep 2005)
4	33	Jul – Sep 2005
6	31	Sep 2005
7	26	Aug – Sep 2005
10	16	not available
11	13	Jun –Sep 2005, Jul – Oct 2006
12	25	Aug – Oct 2005
Permanent		
14	53	-

Water depth in the study pools changed seasonally and dramatically, especially in 2005. Water depths in the permanent pool 14 remained high in winter 2004 and then started to decrease from May to September 2005 (*Figure 2.6*). In late autumn 2005, water level in pool 14 started to increase and remained high persistently until summer 2006 when it started to decline again. Relative change in depths was seasonally similar in all pools. However, most temporary pools dried out completely in late summer 2005.

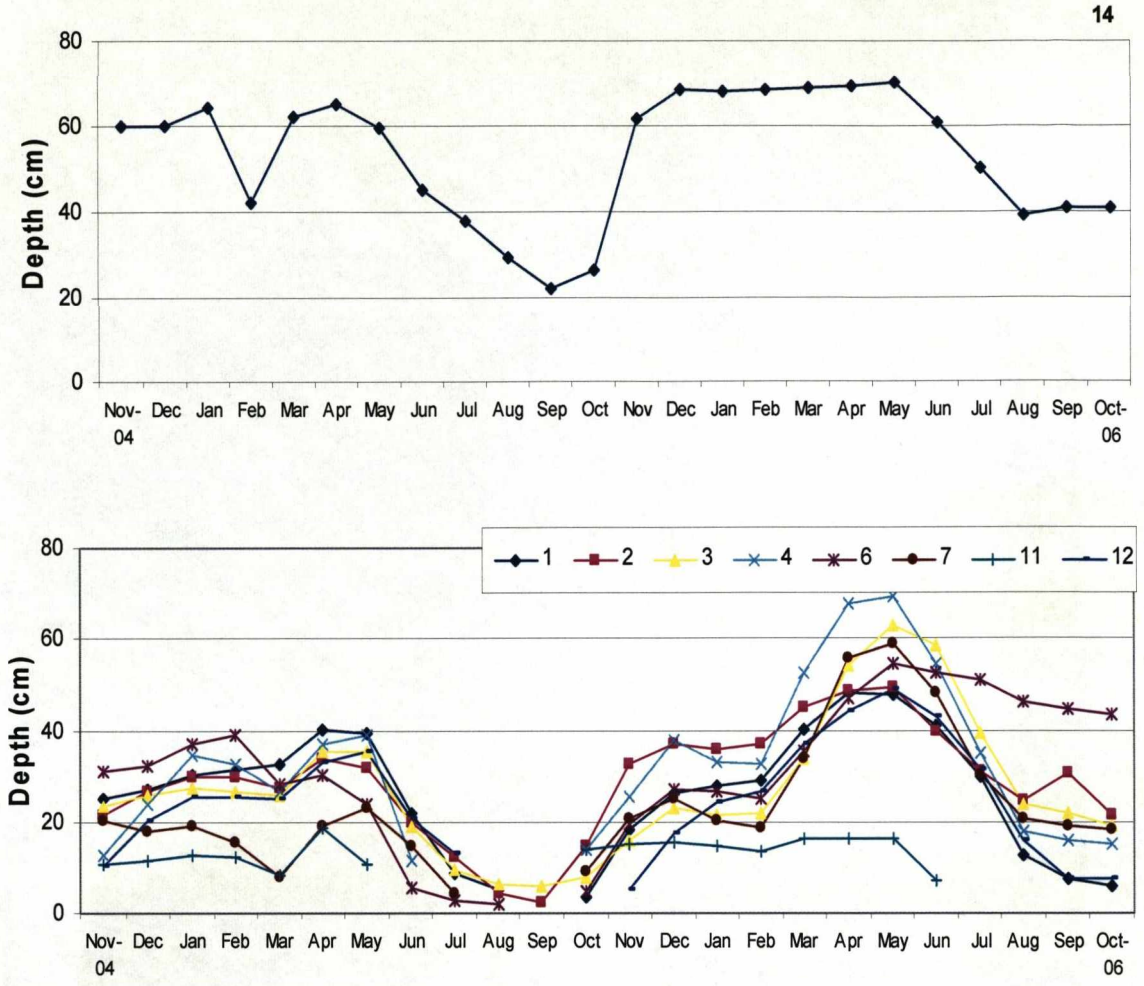


Figure 2.6 Seasonal variation of water levels in a permanent pool (top) and temporary pools (below)

Figure 2.7 demonstrates changes of alkalinity and conductivity values in a permanent pool (14) and temporary pools. Seasonal variations of both alkalinity and conductivity values in a permanent pool were relatively small. In temporary pools, alkalinity values tended to increase in winter and summer and conductivity also increased in summer. Especially during the period before drying out and immediate after refilling with water, alkalinity and conductivity in temporary pools increased (*Figure 2.7*).

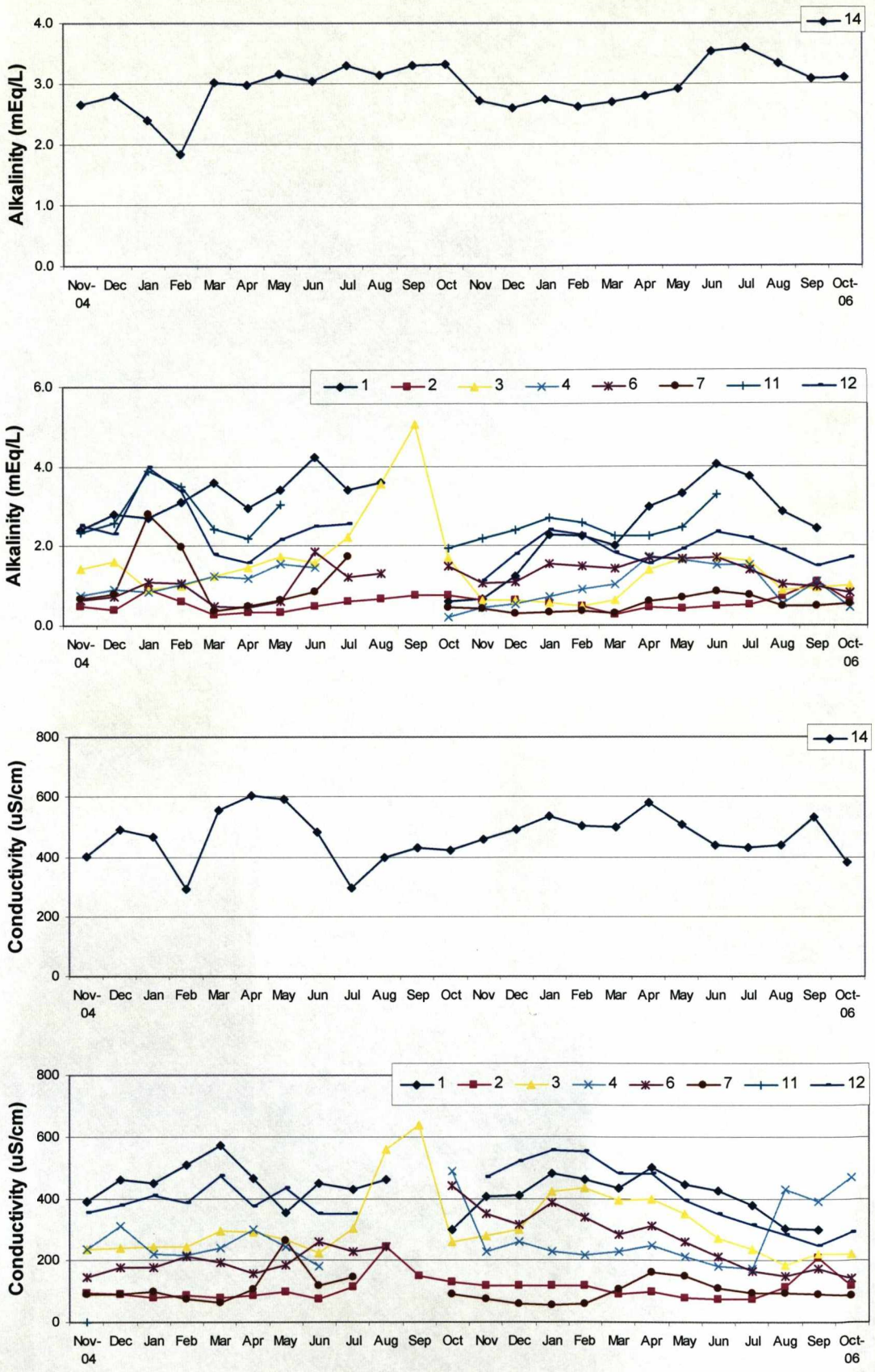


Figure 2.7 Seasonal changes of alkalinity and conductivity in a permanent pool (14) and temporary pools at Brown Moss

In the permanent pool 14, TP increased in summer and decreased gradually in winter (*Figure 2.8*). Similarly, TP concentrations in most temporary pools also tended to increase in summer when water levels also declined and again after the pools were filled by water in early autumn. TN concentrations in the permanent pool were steady throughout most of the year except in winter when TN concentrations increased. In contrast, TN concentrations in temporary pools varied and changed dramatically, particularly in summer before the pools dried out and in early autumn after the pools were submerged.

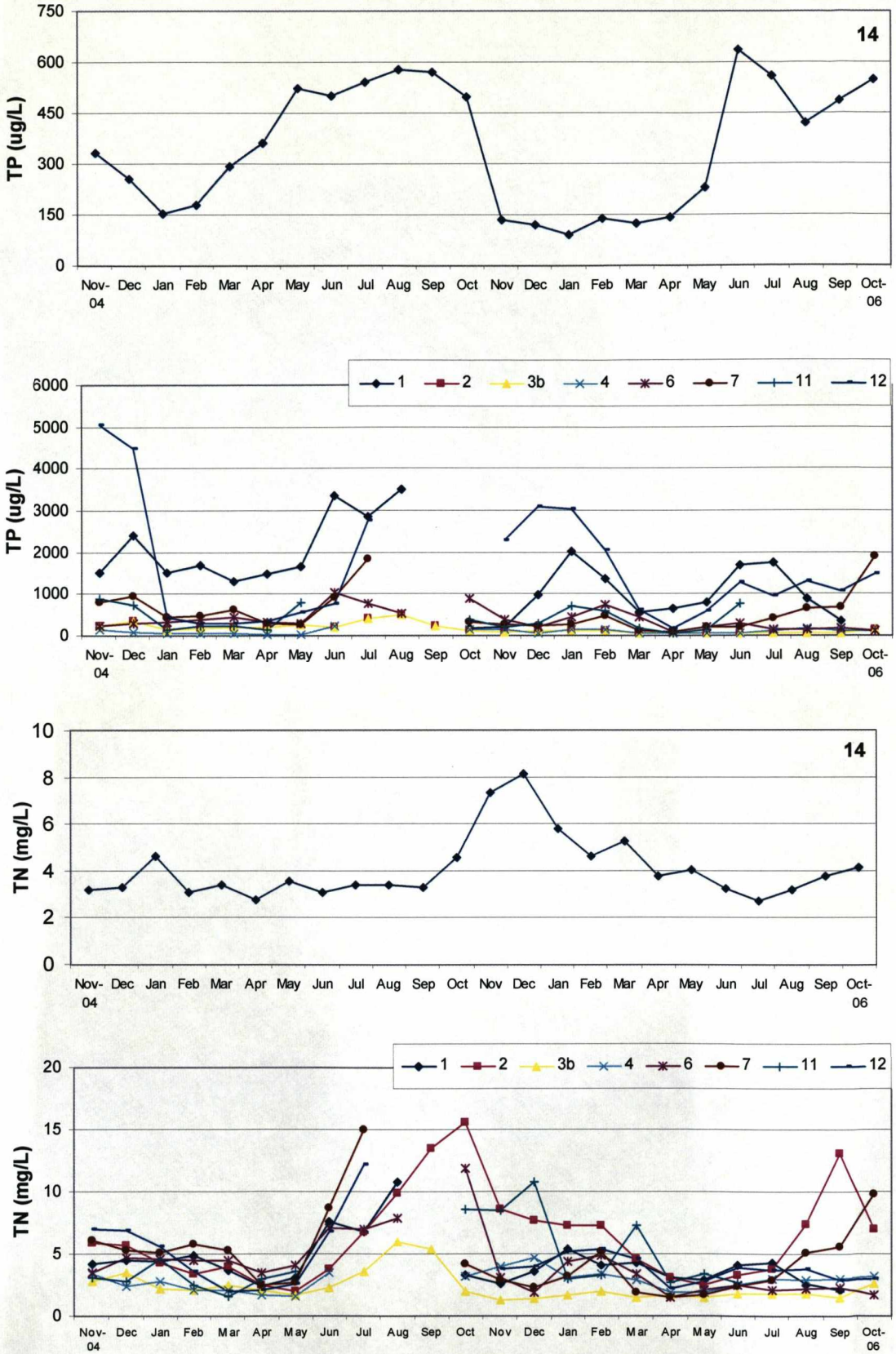


Figure 2.8 Seasonal changes of TP and TN in a permanent pool (14) and temporary pools at Brown Moss

2.5 Discussion

2.5.1 Nutrient status and impact of land use on pools

Alkalinity and conductivity values were relatively high in pools located around the fringe of Brown Moss (pools 1, 10, 11, 12) and pool 14 and this suggests that these pools may receive water from soil or ground sources that has had prolonged contact with soil minerals. Eilers et al. (1989) found that conductivity and alkalinity during a 50-year period in selected northern Wisconsin lakes have increased and increases in such variables show an association with increases in land use development on the lake perimeters. In contrast, pools that had lower values of alkalinity and conductivity may less be affected by development catchment. Values of only 100 $\mu S\ cm^{-1}$ (pools 2 and 7) suggest little modified rain water sources.

The similarity analyses produced different classifications for conductivity/alkalinity and total nutrients. It has often been believed that major ion contents, the most prominent determinants of conductivity, and nutrients are correlated. This may be the case in pristine habitats where soft rocks and glacial drift deposits release more nutrients than igneous catchments, but even then the nutrient levels are not proportionately higher. Natural vegetation conserves nutrients because N and P are generally scarce in available form and terrestrial systems retain them for recycling. Pristine freshwaters have very low concentrations of N and P with no more than 10-50 $\mu g\ P\ L^{-1}$ or 0.5-1.0 mg $N\ L^{-1}$ (Likens et al. 1971; Smith et al. 2003; James et al. 2005). In contrast conductivity may vary from <50 to >1000 $\mu S\ cm^{-1}$.

Where values for nutrients exceed these pristine ranges, there will have been contamination from human activities. All cultivation and grazing removes the conservation mechanisms of natural vegetation. In general animals, including humans, excreta used in crop lands are higher in N than P and in agricultural sources, inorganic nitrogen is more soluble than phosphorus compounds and more easily washed out of soils. Addition of excreta and inorganic substances from human activities combined can lower water quality and leads to problems.

Analyses of the permanent and temporary shallow pools revealed that most pools at Brown Moss are currently suffering from high nutrient concentrations probably from local arable/pasture fields and cottages around the site (*Figure 2.9*). This was consistent with other shallow lakes such as those study by Sondergaard et al. (2005) who found that a majority of the shallow lakes in Denmark were eutrophic and had total phosphorus concentrations above 0.1 mg L^{-1}. In particular, pools 1, 2 and 12 were most polluted compared with other pools. Figure 9 shows that pools 1, 2 and 12 are located close to Croft, Bee Hive and Honey suckle Cottages, respectively, and concentrations of TP in those pools were exceptionally high, whilst N:P ratios were comparatively low (around 3 in pools 1 and 12, 6 in pool 2) suggesting nitrogen was limited. Seepage of septic tank systems and domestic wastewater from cottages is likely to cause such excessive TP concentrations. Several studies have shown that septic tank effluents from cottages can migrate to receiving waters and increase nutrient concentrations (Robertson et al. 1991; Dillon and Molot; 1996). Ptacek et al. (1994) found that a 3 mg P L^{-1} contour migrated horizontally 2.4 m yr^{-1} along one particular path in carbonate-rich sand and also septic tank TP reached the littoral zone in Harp Lake, Ontario, Canada (Wood, 1993). A further examination of this using soil water concentrations is given in Chapter 3.

Apart from being affected by septic tanks, pool 2 also seems to have been affected by farm animal wastes (*Figure 2.10*) as it had the highest concentrations of total nitrogen and ammonia. Several studies have shown that animal waste causes high concentrations of nitrogen in lakes. For example, Ham et al. (1998) and Ham and DeSutter (1999) found that ammonium-N concentrations in animal-waste lagoons were high. Excess manure production on agricultural lands may also create surplus N, which is mobile in many soils and often leached to aquatic ecosystems (Carpenter et al. 1998). Seepage of waste water from a cottage and animal wastes combined may have caused pool 2 to be one of the most polluted water bodies at Brown Moss.

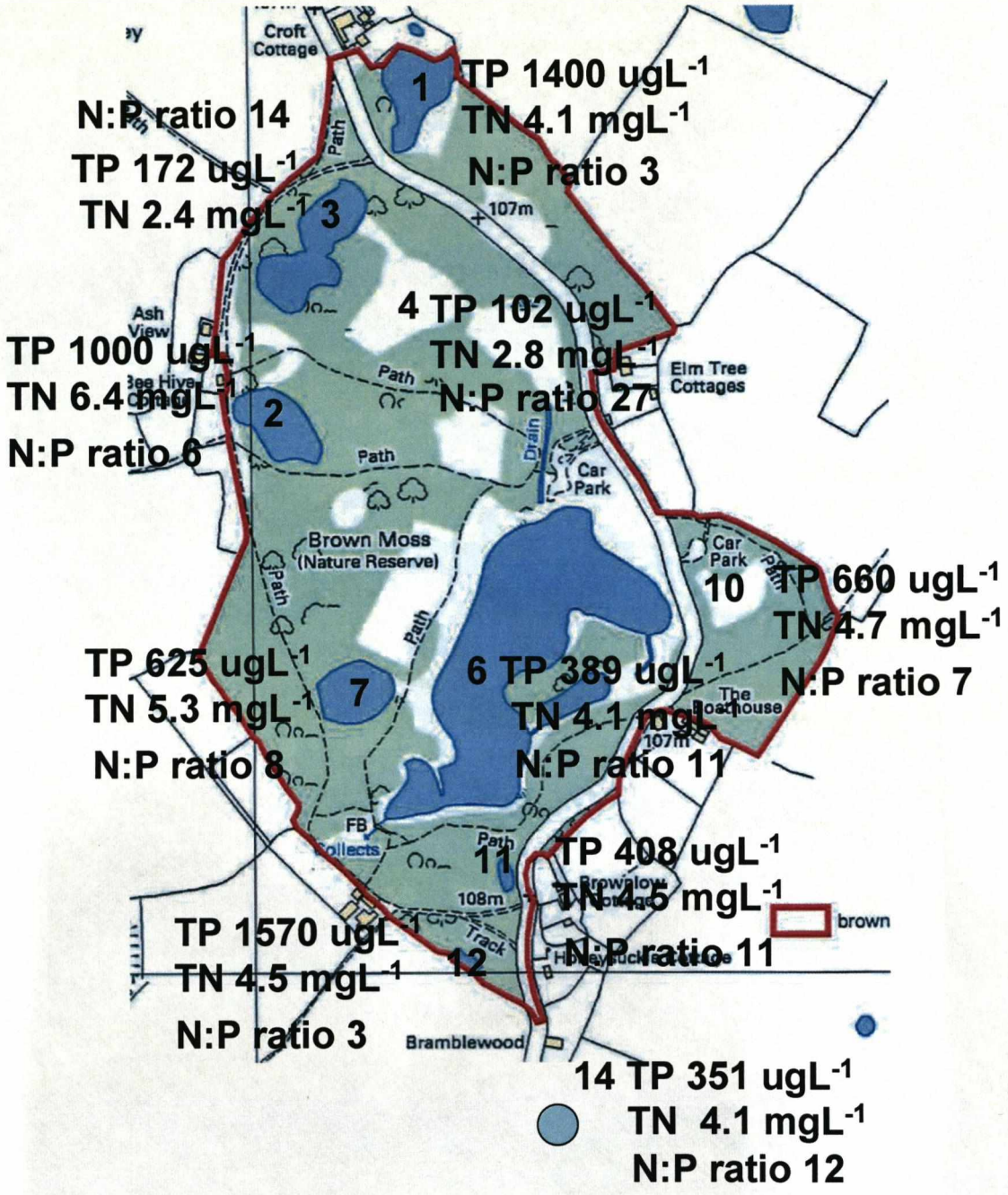


Figure 2.9 Mean concentrations of TP and TN and the N:P ratio in the study pools in and around Brown Moss

Figure 2.10 Land use around pool 2

Other pools located around the edge of Brown Moss also had high concentrations of nitrogen possibly due to the effect of agricultural activities. Pools 11 and 14 were much richer in nitrate than any of the others and N:P ratios ranged to around 11 and 12, respectively (*Figure 2.9*). Fertilizers and manure are applied to crops around Brown Moss during the growing season and with continuous application of nutrients, the storage capacity of the soil may be quickly exceeded. As a result significant amounts of nutrients from the surrounding agricultural areas can get into the pools through surface run-off and sub-surface percolation following heavy rain and cause eutrophication problems (Kortelainen and Saukkonen, 1998). Berka et al. (2001) reported that agricultural intensification is leading to significant problems in watersheds of the Lower Fraser Valley in Washington State and British Columbia, Canada and is impacting water quality, particularly during the wet winter season resulting in low dissolved oxygen, high ammonia and nitrate levels. Therefore, non point source pollution is a major source of nutrients to surface waters and primarily results from agriculture (Carpenter et al. 1998).

Chlorophyll a analysis shows that pools 7, 10 and 14 were dominated by phytoplankton with high concentrations of chlorophyll a. In pool 7, in particular, phytoplankton caused turbidity as significant correlation between TSS and chlorophyll a concentration was found. Being open waters without shading from trees may help promote the growth of phytoplankton since light is needed for photosynthesis. By comparison with pools 7 and 10, pools 11 and 12 had lower concentrations of chlorophyll a and this may be the result of shading effects from over

grown scrub and tress around the pools (*Figure 2.2*) that decrease light availability, thus limiting phytoplankton development. Phlips et al. (1995a, b) hypothesized that light limitation may prevent phytoplankton standing crops from reaching the potential biomass provided by the prevailing available nutrients. In Lakes Chapala (Limon et al, 1989) and Chad, Mexico phytoplankton standing crops are unexpectedly low due to the severe restriction of light availability (Brylinsky and Mann, 1973).

Furthermore, high concentrations of chlorophyll a in pools 7, 10 and 14 may be related to the absence of submerged macrophytes (*Figure 2.2*) compared with pools 3 and 4 dominated by water plants and low chlorophyll a concentrations. This is because uptake of nutrients by aquatic plants in the pool may keep available nutrient concentrations low, thus preventing phytoplankton from developing. Kufel and Ozimek (1994) explained that macrophytes and their associated epiphytes take up and store nutrients which are not available for phytoplankton to use. Sondergaard and Moss (1998) also found that the presence of macrophytes may influence nutrient cycling and decreased availability in nitrogen and sometimes phosphorus may be expected in most cases.

The low concentrations of chlorophyll a were detected in pools 3 and 4. This may be explained by phytoplankton feeders associated with plants as there may be higher numbers of zooplankton in pools 3 and 4, that keep water clear, than in pools 7 and 10 lacking submerged macrophytes. Sondergaard and Moss (1998) and Timms and Moss (1984) explained that a lower phytoplankton biomass is found in the presence compared with the absence of macrophytes. This is because zooplankters generally occur in greater numbers inside and around macrophyte beds and zooplankton such as *Daphnia* can prevent phytoplankton biomass from responding to nutrients (Sarnelle, 1992). In addition, lower concentrations of chlorophyll a in pools 3 and 4 could be the result of low nutrient concentrations that limit growth of phytoplankton.

Pools 3 and 4 represented water bodies less disturbed by anthropogenic activities because they had lower concentrations of nitrogen and phosphorus compared with other pools around Brown Moss. Pools 3 and 4 are situated in inner areas of the site and are surrounded by woodland and this may help protect the pools from becoming more affected by nutrient additions from external sources. Nutrients such as nitrogen

can be removed from the systems through denitrification (Sondergaard et al. 2005). Current nutrients in pools 3 and 4 are believed to derive from accumulated sediment organic matter, which that can be a significant source of recycled nutrients for water column productivity (Logan and Langmore, 2008). A study of internal nutrient loading in Lake Ringsjön showed that amounts of nutrients are stored in sediments especially in summer when external additions are minimal (Granéli, 1999). An experiment concerning release of nutrients from the sediment at Brown Moss is presented in Chapter 3 and 4.

2.5.2 Environmental stress and changes of water chemistry in permanent and temporary ponds

Water levels in permanent and temporary pools at Brown Moss changed seasonally. They increased in winter and decreased in summer. This suggested that most pools received their water from winter rain and surface runoff, and lack of rainfall and high rates of evaporation over rainfall in summer caused a decrease of water levels (*Figure 2.11*). Collinson et al. (1995) also mentioned that many temporary ponds rely on direct precipitation and climate. In some years many pools dried out in summer and Whild (2003) suggested that drying out of the pools at Brown Moss is caused by several possible causes, including climate change, siltation of the pools and increased rate of evapotranspiration. Water tables in the area fluctuate greatly in each season (Chapter 3).

Figure 2.11 Drying out of pools 4 (left) and 12 (right) in summer 2005

In summer, drying out of temporary pools may cause environmental stress on aquatic organisms. Williams et al. (1999) maintain that water levels need to remain stable for a pond to be healthy and drying out is disastrous for wildlife and water level fluctuation should be minimised. Williams and Feltmate (1992) also reported that drying out of ponds in a drought year will have a catastrophic effect on the invertebrate community. Temporary ponds in Oxfordshire, England support relatively few aquatic macroinvertebrates when compared with more permanent sites (Collinson et al. 1995). To survive during the dry period in temporary ponds, animals have to develop various adaptations such as building burrows or in many cases fauna such as insects have to have larval stages that are aquatic and metamorphosis has to be completed before the habitat dries (Kenk, 1949). If aquatic organisms are not well adapted in temporary habitats with dramatic fluctuation of water levels, they will disappear from the pools. On the contrary, some aquatic communities may benefit from drying out because some taxa may be especially adapted to dry phase, others may be able to colonise rapidly.

However, temporary drying out may also be beneficial to the pools. Collinson et al. (1995) stated that drying out leads to the remineralisation of nutrients, thus promoting productivity, while shallowness of temporary ponds enables them to warm up quickly in spring encouraging growth in species which have high thermal coefficients for growth. In addition, lack of large predators such as fish due to winter fish kill and summer dry out may benefit open water invertebrate species, thus increasing faunal biodiversity (Collinson et al. 1995; Sondergaard et al. 2005). And lastly, the drying of temporary pools seems likely to result in rapid oxidation of organic matter during the dry phase, which substantially reduces the rate of sediment accumulation.

Furthermore, during December to February, most temporary pools were covered by ice for at least a few days and this may potentially cause environmental stress through winter kill to aquatic animals. Fish were absent from temporary pools but permanent pool 14 is the only pool in this study that is a fish habitat. In small shallow pools, water may become frozen throughout water column during winter months and this can kill freshwater animals. In bigger pools, an ice cover may cause depletion of dissolved oxygen and this is likely to affect bottom dwelling animals leading to death (*Figure 2.12*). Greenbank (1945) found a steady decline of dissolved oxygen during the long

period of snow cover in a shallow eutrophic Green Lake, America. Several studies have also shown that lowered dissolved oxygen tension due to ice cover in winter causes the death of oxygen-consuming animals such as fish (Cooper and Washburn, 1949; Welch & Bergmann, 1985; Jeppesen et al. 2001). Various noxious gases such as carbon dioxide, methane, hydrogen sulphide, and possibly ammonia from processes of decay may also be likely to cause mortality of fish during an ice cover (Greenbank, 1945).

Figure 2.12 Pool 6 was entirely covered by ice in December 2005

Fluctuation of water levels may have a strong impact on water chemistry. Total nitrogen in temporary ponds at Brown Moss increased dramatically before and after the period of drying out. Prat and Daroca (1983) found that water level fluctuations are very important for the trophic status of water bodies and a period of low water level may result in eutrophication. Total phosphorus in permanent and temporary pools behaved similarly in that it tended to increase in summer. High concentrations of phosphorus in summer are likely due to nutrient release from the sediment (Bostrom and Pettersson, 1982; Stephen et al. 1997; Sondergaard, 1999).

Furthermore, because pools are shallow and small, they are very vulnerable to changes and environmental degradation. Low water volumes of small ponds relative to catchment sizes may mean that they receive larger nutrient loads per unit volume

and also have less potential to dilute pollutants compared with deeper and bigger water bodies (Williams et al. 1999). This is in agreement with Collinson et al. (1995) who stated that quite small changes to the hydrology of a region can potentially lead to major changes in the hydrology of temporary ponds in that area and that they are greatly at risk to polluted surface runoff and easily damaged (Serrano and Serrano, 1995).

At Brown Moss, many small and shallow pools are becoming shallower (*Figure 2.13*) and being filled in. Natural succession caused by continuous accumulation of eroded soil and leaf deposition seems to be a significant cause. Especially in pools 11 and 12, trees overhang most areas of the water surface and thick layers of leaf litter are found in the pool. Shropshire County Council is involved in the management of the surrounding scrub and trees around the pools to prevent them from encroaching on the pools. The vegetation around some of the pools is regularly cut and removed to reduce effects of shading, leaf deposition and silting (Edwards, 2007).

The problem of small pools becoming shallower also occurs across the United Kingdom as the small size of many ponds in particular means that they can easily be filled in (Collinson et al. 1995). A recent survey showed that during the 20th century large numbers of high quality ponds have been destroyed and several ponds that existed at the beginning of the 20th century are now gone (Williams et al. 1999; Biggs et al. 2000; Wood et al. 2003). The main threats that have significantly reduced the number of ponds include natural succession, agricultural land drainage and developments for urban housing, industry or transport infrastructure (Boothby and Hull, 1997).

Figure 2.13 Accumulation of sediments in pool 6 (left) and pool 11 being filled in by leaf litter (right) in summer 2005

Loss of shallow and small freshwater habitats due to filling up from such materials may magnify and cause ecological changes and problems, predominantly in summer and autumn. Several shallow pools at Brown Moss are aquatic habitats of great crested newts (*Triturus cristatus*) (Edwards, 2007) and therefore losses of water bodies may directly affect their population. As a pond fills with sediment and becomes progressively shallower, the community it supports will also gradually change (Williams et al. 1999). Over the past 50 years, there has been a big decline in protected species such as great crested newts partly due to a loss of aquatic habitat (fishless ponds where there are lots of weeds) (BBC News, 2008). To compound the problems of habitat loss, newt larvae require open water and as a result are particularly vulnerable to predation by fish (Buckley, 2001). At Brown Moss, Shropshire County Council has drawn a management plan to maintain a long-term viable population of great crested newts, at favourable status, across the site. This involves the management of the pools to ensure that there are suitable open areas, marginal cover and sufficient water levels to enable breeding (Edwards, 2007).

In conclusion, although Brown Moss is a protected area, most pools are facing problems as water quality of all study pools is low with excessive concentrations of nutrients. Land use around Brown Moss and internal loading appeared to be important factors that degrade water quality in the pools. Furthermore, changes of water levels had an impact on concentrations of nutrients especially during the period of before

and after drying out. Temporary and shallow pools also appeared to have greater fluctuations of water chemistry than in a permanent pool. Lastly, natural succession is likely to cause pools to become shallower and this may lead to losses of aquatic habitats and to reduction of ecological importance of the site.

Chapter 3 Water and nutrient budget of Brown Moss

Chapter 3
Water and nutrient budget of Brown Moss

3.1 Introduction

In lake ecosystems, nutrients play a crucial role in primary production and trophic status as nutrients are essential for photoautotrophic producers such as phytoplankton and plants. Nitrogen and phosphorus are especially important (Schindler, 1977; Portnoy, 1990; Burkart and James, 1999; Sondergaard et al. 2001). In shallow lakes, if nutrients are supplied in modest quantities, growth and productivity of autotrophs are considered beneficial. However, when nutrients are available in excessive amounts, they can increase phytoplankton biomass, degrade water quality and cause undesired biological changes that have adverse impacts on both animals and human beings (Brönmark and Weisner, 1992; Moss et al. 1998; Padilla et al. 2006).

Where do nutrients in lakes come from? Nutrients ultimately come from the land and activities in the watershed, sometimes, as nitrate and ammonium, and from the atmosphere, and may also be secondarily delivered after storage in the sediments (*Figure 3.1*). They can be divided into non-point, point and internal sources (*Figure 3.2*). Non-point sources are diffuse sources stemming from cultivated areas and livestock farms. Non-point source contamination starts when rain falls on the ground and runoff moves and washes out nutrients from the land and finally transports them to lakes. As a result, too much animal manure and chemical fertilizer on agricultural lands can increase the nutrient contamination of the run off (Gallimore et al. 1999).

Nutrients transported through soil and groundwater over the surface after rain are of concern. Verchot et al. (1997) stated that subsurface flow often represents the major pathway for movement of dissolved nutrients such as nitrate from cropland. As contaminated water seeps through soil, it carries nutrients and as it moves deeper, it can contaminate groundwater. Since many lakes receive water from ground water, nutrients in ground water may affect the quality of lake.

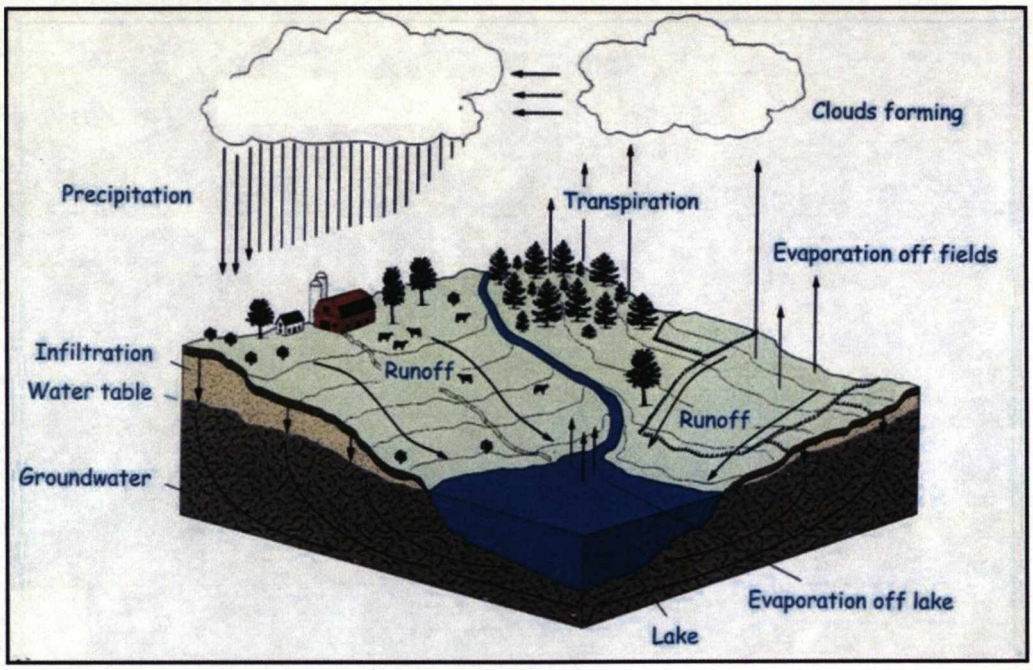


Figure 3.1 Watershed and hydrological cycle of lakes
(source; Michigan State University)

Atmospheric deposition is also a non-point source of pollution (USEPA, 2007) and can be a significant source of nutrient input to lakes (Porter, 1975; Jassby et al. 1994; Aillery et al. 2005). Junge and Werby (1958) stated that the concentration in rain water reflects to a large extent the concentration of the same constituent in the air and is, therefore, of special importance.

Volatilization of nitrogen-based gases from agricultural land, and other airborne pollutants, generated by human activities, can supply nutrients (N) to ecosystems from atmospheric deposition (Woodmansee, 1978; Tilman et al. 1998). Moss (1998) noted that nitrate in rain may be eutrophicating, especially from oxidation of ammonia volatilized from animal wastes. In the United States, large quantities of ammonia are emitted from animal operations each year, making up to 50 to 70 percent of ammonia emissions from all sources (National Research Council, 2003).

Nutrients in lakes also come from point sources, where pollution can be traced back to precise sources of origin (*Figure 3.2*). Discharges from industry and domestic sewage are examples of point source pollution. Liquid and semi liquid wastes from households include washing and laundry water and cooking and human wastes. Domestic wastes, especially, contain high concentrations of ammonium, nitrate and phosphate. When they leak from sewer pipes or septic tanks or are directly discharged into water bodies, they may create major impacts and change in aquatic ecosystems.

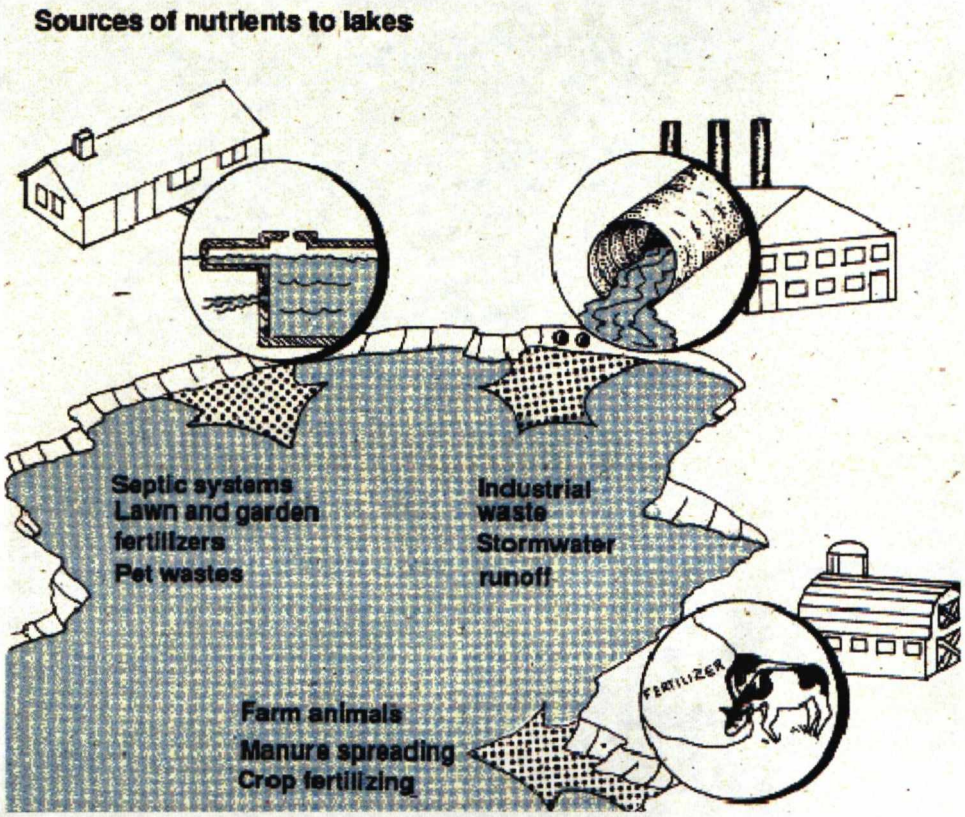


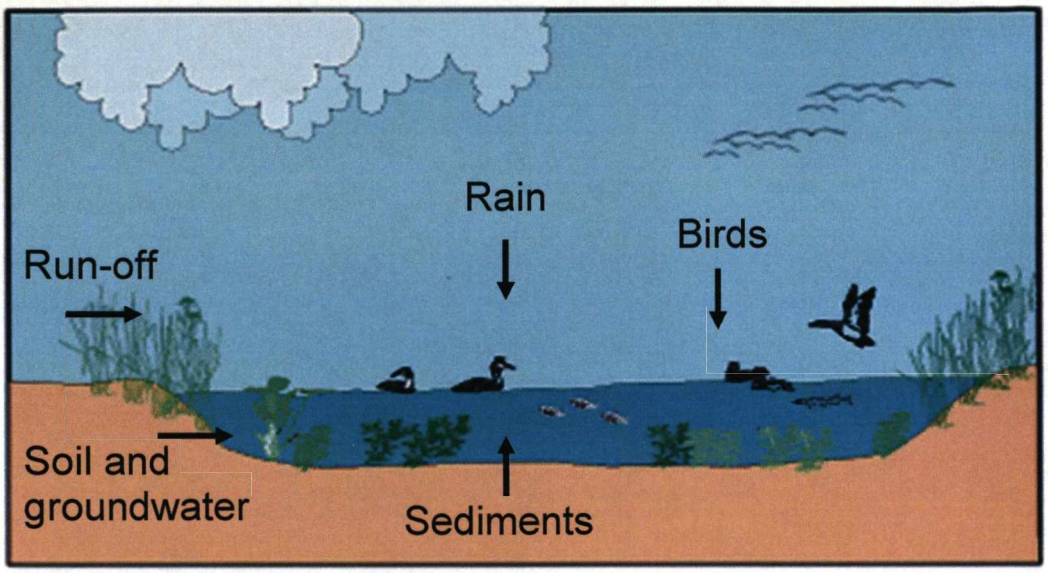
Figure 3.2 External nutrient loading to lakes comes from point and non point sources. (source; http://www.ecy.wa.gov)

Fauna such as hippopotami and birds, in the tropics, directly influence the supply of nutrients (*Figure 3.2*) and can be an important external source to the lake (Andersson, et al. 1988). Addition of nutrients by waterfowl is called "guanotrophication" (Leentvaar, 1967; Moss and Leah, 1982; Gere and Andrikovics, 1992; Irvine et al. 1993; Kitchell et al. 1999). Faeces, excreted by birds, contain nitrogen and phosphorus. When waterfowl gather in lakes, high levels of bird excretion and defaecation may lower water quality and promote the growth of phytoplankton,

leading to eutrophication problems (Marion et al. 1994; Post el al. 1998; Manny et al. 1994; Vanni, 2002; Mukhergee and Borad, 2001).

Apart from external sources of nutrient loading to lakes as mentioned above, internal sources such as nutrient release from the sediments also play a crucial part. As described by Pettersson (1998), release of nutrients from particles and interstitial water in the sediment is called internal loading. Internal loading is regarded as an important source of nutrients in lakes since nutrients can be released to the water column through physical, chemical and biological mechanisms (Jensen and Andersen, 1992; Sondergaard et al. 2003; Nowlin et al. 2005). Phosphorus release from the sediments may substantially increase the bioavailable phosphorus pool and consequently also the algal biomass (Knuutila et al. 1994). Ultimately, however, the source of nutrients in the sediment is that of external loading.

Lakes can be vulnerable to degradation and eutrophication if excessive nutrients enter from external and internal sources combined. To reduce and control excessive nutrient loading to lakes, estimating the amounts of nutrient input from potential sources within its catchment area is required in order to assess the biological response of nutrient loading. Accordingly, in this chapter, I have attempted to create a model to quantify nutrient loading and budgets of total nitrogen and phosphorus coming from various potential sources into pool 6 (number 6) at Brown Moss. Nutrient loading (L) refers to the total amount of nutrients (nitrogen and phosphorus) entering water bodies over a given time and is measured, for example, as kilogramme of nitrogen per year (kg-N yr^{-1}) while concentration refers to amounts of nutrients per unit volume of water, such as milligram per litre (mg L^{-1}). The relationship between nutrient loading and concentration can vary and depends on the flow, the amounts of nutrients entering, the volume of water in the pool, and drainage basin characteristics (Vollenweider, 1968, 1974; Maryland Department of Natural Resource, 2007).



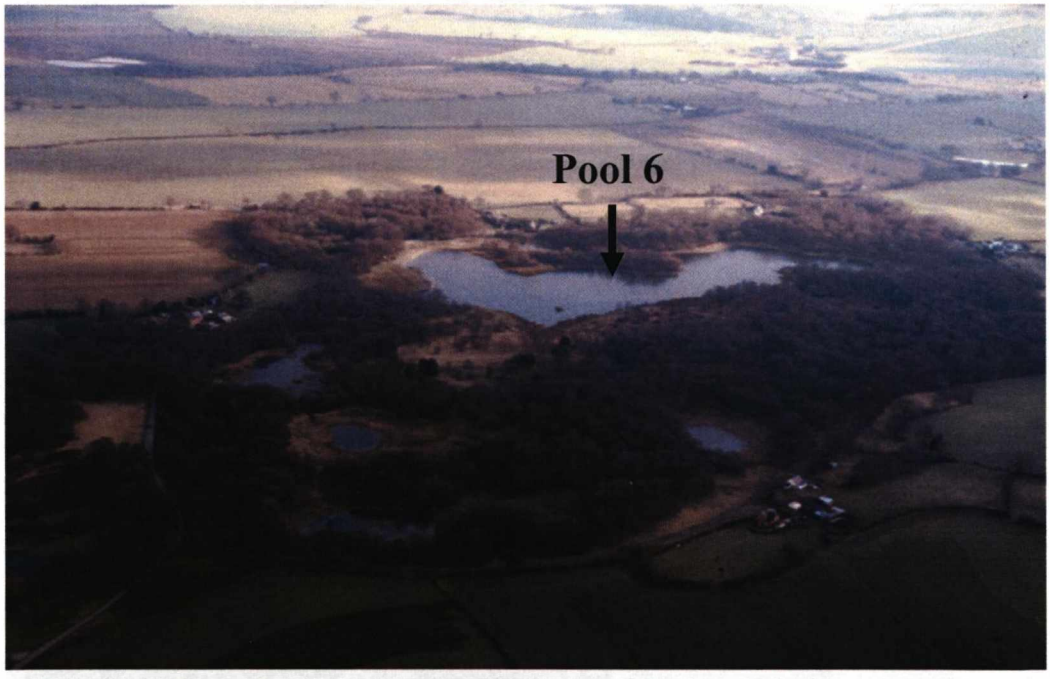


Figure 3.3 External and internal inputs of nutrients in lakes (source : Scheffer, 1999) (above) and the study pool 6 at Brown Moss (below)

Estimation of nutrient budget in pool 6 (*Figure 3.3*) at Brown Moss was carried out. Pool 6 is isolated and there are no direct river/stream fluxes. Accordingly, the components of external and internal nutrient loading to the pool included inputs from birds, from surrounding areas through soil and ground water, from rain water and from release of nutrients from sediments (*Figure 3.3*). Data obtained led to assessment of trophic status of the pool and a suitable plan for reduction of nutrient loading if desired.

3.2 Methods

3.2.1 Waterfowl

I studied waterfowl in pool 6 to estimate nutrient loading by birds. Birds present near, and in the pool, were counted by species every two weeks (Bibby, 1992) using binoculars (10×50 wide angle). I counted the number of birds around 13.30 pm from a place where I could see the whole pool clearly and did not disturb birds while counting. Birds were also counted by species every hour from dawn to dusk in December 2004, February 2005, April 2005, June 2005, August 2005 and October 2005 to study diurnal change and to establish how representative the more frequent count at 13.30 hours might be. The identification of waterfowl was based on experts and bird guide books (Johnson, 2003; Flegg and Hosking, 1990).

Bird excretion rates were assessed by the calculation of amounts of nutrients produced by different bird species, mainly based on literature survey. According to Manny (1994), the daily nutrient load to Wintergreen Lake in Michigan, U.S.A. by migrant Canada geese was on average 1,570 mg TN and 490 mg TP per individual per day. He assumed that nutrients added by ducks, for example, were proportional to their body weight relative to that of an average goose. In this study, nutrient inputs by other bird groups to pool 6 in Brown Moss were calculated based on this relationship. I also used the amount of TP produced by black-headed gulls, which is 38 mg per day, to estimate nutrient inputs by other birds (Gould and Fletcher (1978) in Moss and Leah (1982)). I determined nutrients added by water birds from a log/log relationship between body weight and a curve determined from geese and black headed gulls. I used a log/log relationship to transform data and to minimise mathematical error between the highest and lowest values.

Monthly nutrient load from birds to the pool was based on three main factors;

1) Nutrient loading rates (TP and TN) from different bird species based on literature survey and estimation.

2) Monthly numbers of waterfowl recorded in the pool were estimated by bird counts every two weeks and then I plotted a graph and the area under graphs is the total numbers of birds estimated.

3) Area of the pool

Annual nutrient (TP and TN) loading to pool 6 was estimated as;

$$= \frac{\text{Numbers of waterfowl} \times \text{nutrient excretion per bird (mg yr}^{-1}\text{)}}{\text{Area of pool (m}^2\text{)}}$$

Unit = mg m^{-2} yr^{-1}

3.2.2 Ground water and run off

Shallow soil and groundwater flowing from land was collected by ten tubes (lysimeters of 10 cm diameter×1.50m deep) installed in two lines N/S and E/W across the area of Brown Moss nature reserve in January 2005 (*Figure 3.4 and 3.5*).

Figure 3.4 Boreholes at Brown Moss

I used a barrel auger to create boreholes and then I sank each tube approximately 1.0 metre in the ground, leaving 0.50 metre above ground surface. Afterwards, surrounding holes were backfilled with native soil and the tube tops were covered with rubber stoppers to avoid disturbance and contamination from outside. Before sampling, the depth of water in the boreholes was measured with a metre rule. Afterwards approximately 300 ml of groundwater was taken, using a small plastic bottle with long stick handle and transferred to acid washed glass bottles. Conductivity and pH were measured on site using a Hanna HI 9812 pH meter. Water samples were brought back to the laboratory for chemical analysis (see section 3.2.5). Water samples were taken every two weeks from February 2005 to October 2006.

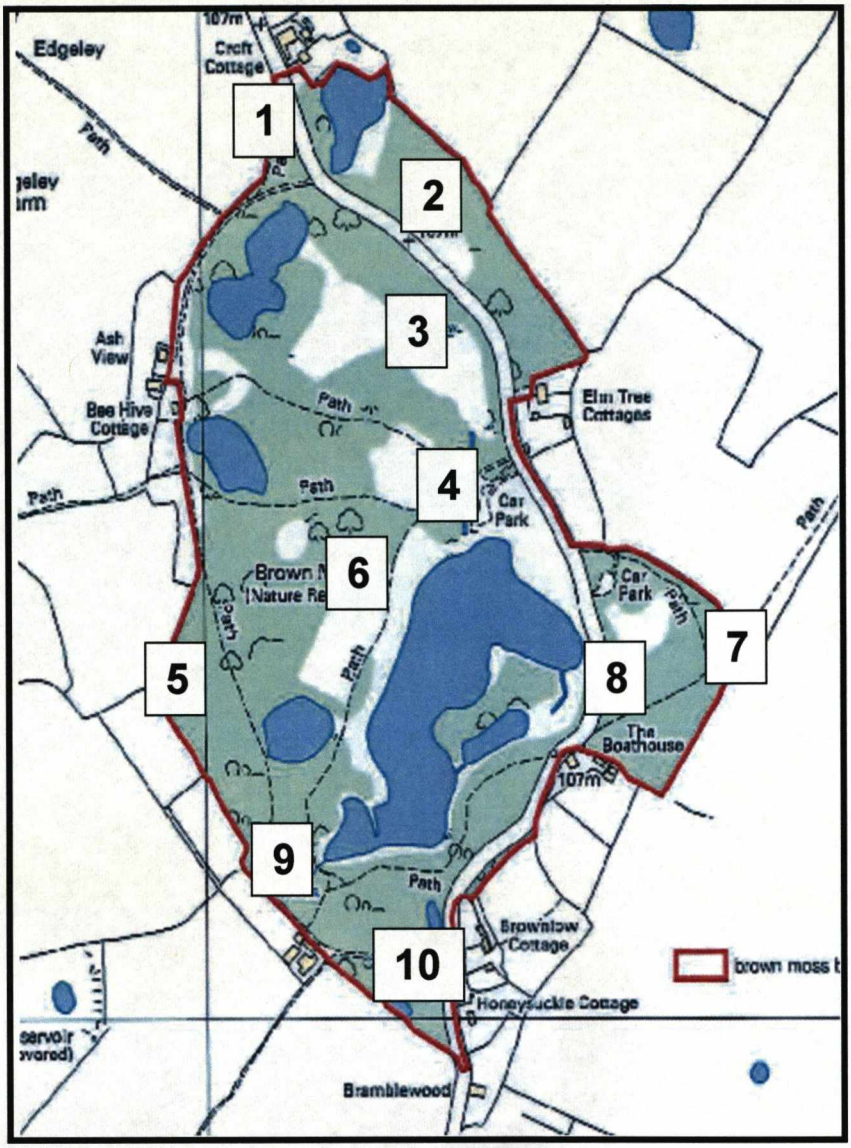


Figure 3.5 Map of boreholes installed across the site

Estimation of nutrient loading by shallow ground water to the pool was calculated based on volume of water entering the pool, mean concentration of plant nutrients from boreholes and area of catchment. I calculated the amount of water entering the pool or effective rainfall by using average annual precipitation minus evaporation (Moss et al. 1998). The average annual rainfall and evaporation at Brown Moss were 0.70 m yr^{-1} and 0.57 m yr^{-1}, respectively (Environmental Consultancy, 2001). However, there are some difficulties in estimating water entering the pool as this model did not include water that may lose through deeper ground water, or may be used by plants or may be stored in soil particles.

To obtain the catchment area of pool 6, I considered values of conservative ions (alkalinity, conductivity and winter TN) of groundwater obtained from boreholes around the site compared with those of the water in the pool to generate the likely catchment area. The conservative ions are those whose concentrations are not greatly affected by biological changes, and which are highly soluble.

The amounts of TP and TN entry were then calculated based on a model presented below;

$$= \text{Average TP and TN concentrations from appropriate boreholes (mg m}^{-3}) \times \frac{\text{(Precipitation - Evaporation) (m yr}^{-1}) \times \text{catchment area (m}^2)}{\text{Area of the pool (m}^2)}$$

Unit = $mg\ m^{-2}\ yr^{-1}$

3.2.3 Rain water

Due to convenience and limitations of chemical analysis, I collected rain water at the University of Liverpool to estimate nutrient loading from rain to the pool. It is safe to assume that rain collected in Liverpool represents similar nutrient concentration as that falling at the study site, although it is expected that nitrate concentrations from rain water collected in Liverpool may be slightly higher than those at the study site due to the impact of fuel combustion from cars. The study site is about 1 hour and 20 minutes by car from Liverpool. Samples of rain water were collected on a regular basis, usually once a month, by using a plastic rainfall collector (40×57×19 cm), situated above ground level, outside the BioSciences building, School of Biological Sciences, the University of Liverpool. Approximately 300 ml. of rain sample was transferred and kept in glass bottles to be filtered and analysed for nutrients (see section 3.2.5). Rain water samples were collected from January to December 2006.

To estimate nutrient loading by rainfall to pool 6 per square metre, I multiplied concentrations by the average rainfall and the area (Migon and Sandroni, 1999) as shown below;

$$= \text{Average annual rainfall (m yr}^{-1}) \times 1\ \text{m}^2 \times \text{average concentrations of TP and TN in rain water (mg m}^{-3})$$

$$\text{Unit} = \text{mg m}^{-2}\,\text{yr}^{-1}$$

3.2.4 Nutrient release from the sediment

Three locations (A, B and C) in pool 6 were used to measure release of nutrients from the sediment every month from November 2004 until October 2006 with triple replication (*Figure 3.6*). I did not include results of nutrient release obtained from sites D and E because D and E (see Chapter 4) are not connected to pool 6 (A, B and C) throughout the year and their properties were different from those in pool 6 (Chapter 4).

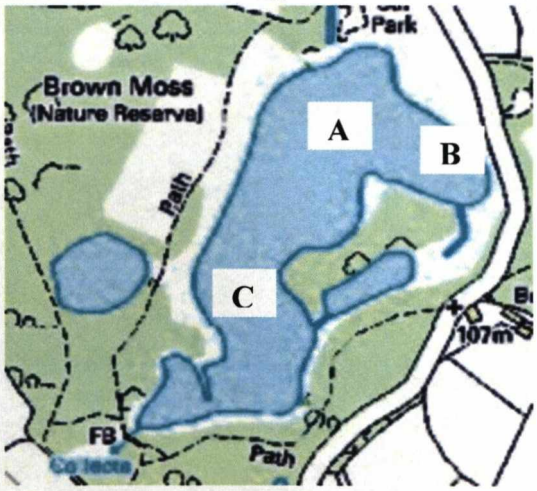


Figure 3.6 Sampling station

I used plastic core tubes (length 50 cm diameter 6.9 cm) to collect the sediment. The sediment and water were taken by coring tubes that were pressed into the pool bed to a depth of approximately 10-15 cm. Corers were then sealed bottom and top with rubber bungs and kept in darkness in square tubes and brought to the laboratory at the University of Liverpool (*Figure 3.7*) for monitoring release rates of nutrients over 24 hours.

Figure 3.7 Plastic core samples were used to study nutrient release from the sediment

After return to the lab, approximately 200 ml of water from each sample was pumped from about 5 cm above the sediment surface through a plastic tube and the water was kept in glass bottles to be filtered and analysed (see section 3.2.5). Afterwards, core samples were kept in square tubes to maintain darkness and incubated for 24 hours in a water bath located in an open area outside the building at ambient temperature, which mimicked the pool environment (*Figure 3.8*). The following day, after 24 hours, 200 ml of overlying water was again taken from the same cores, using a hand pump and water samples were filtered and analysed for nutrients. The volume of water left in the samples was finally measured and nutrient release rates were estimated.

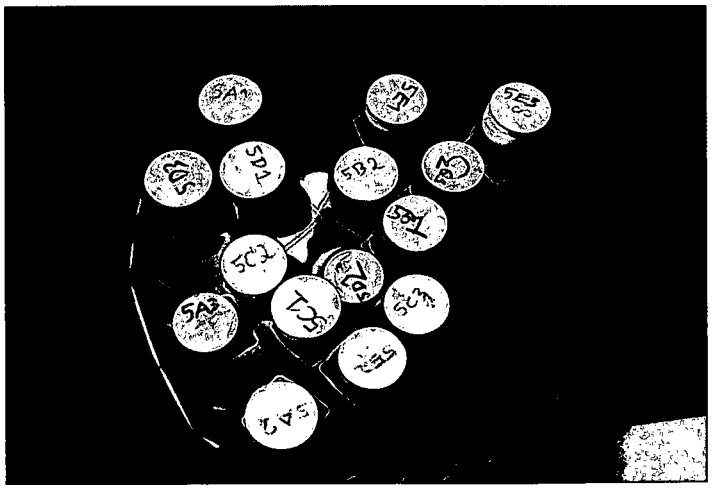


Figure 3.8 Plastic core samples were incubated in an open water tank for 24 hours

Estimation of nutrient release from the sediment is based on the difference between concentrations of nutrients measured on day 1 compared with the concentrations on day 2 (after 24 hours). I measured volume of the water on day 2 to convert concentrations to actual amounts on an aerial basis.

Therefore, to measure rates of nutrient release from the sediment, I used the equation shown below;

= ((Conc. of nutrients Day 2 – Conc. of nutrients Day 1 (mg $L^{-1}day^{-1}$))×Volume of water (L)) × 470.87 (m^{-2})

Unit = (mg m^{-2} day^{-1})

I then multiplied average amounts of nutrients released per day by 365 to give yearly rates of nutrient release expressed as mg m^{-2} $year^{-1}$.

470.87 is an area conversion factor (1 m^2 divided by surface area of the core) that is used for converting the amount of nutrients in the core tube (52mm diameter) to the value for 1 square metre.

3.2.5 Water chemistry and statistical analysis

I collected water samples from pool 6 for two years from November 2004 to October 2006. The water samples collected from pool 6, ground water, rain water, and corers (nutrient release) were filtered through Whatman glass microfibre filters (GF/C, 47 mm). The filtered water was subsequently analysed for soluble reactive phosphorus (SRP, $\mu g\ L^{-1}$), nitrate nitrogen (NO_3^--N, $mg\ L^{-1}$) and ammonium nitrogen (NH_4^+-N, $\mu g\ L^{-1}$), based on Mackereth et al. (1989). Total nitrogen (TN, $mg\ L^{-1}$) and total phosphorus (TP, $\mu g\ L^{-1}$) were measured by methods of Johnes and Heathwaite (1992) using unfiltered water. I also measured pH and conductivity ($\mu S\ cm^{-1}$) using a Hanna HI 9812 pH meter and alkalinity ($mEq\ L^{-1}$) using titration against standard acid to an end point of pH 4.5 (Mackereth et al. 1989) within a few hours after collection.

Descriptive statistics were used to summarise data. I applied Pearson correlation coefficients (*r*) to study correlation between numbers of birds and concentrations of ammonium nitrogen in the pool. For groundwater, I used regression analysis as well as correlation coefficients to determine the correlations among variables. For Bray-Curtis analysis, PRIMER 6.0 was also used to analyse similarity and difference among chemical data of groundwater. Tukey honestly significant difference (HSD) test and Pearson correlation coefficients (*r*) were applied to rain water data and nutrient release from the sediment. And lastly, I used Pearson correlation coefficients among different chemical variables measured from pool 6. SPSS 14.0 for Windows was used for analysis of Pearson correlation coefficients and Tukey HSD, and EXCEL was used to perform regression analysis.

3.3 Results

3.3.1 Waterfowl

Thirteen species of waterfowl were observed in pool 6 during the study period and were categorised into six groups, based on their body weight and sizes. Main residential species of waterfowl recorded throughout the year were mallard whereas Canada geese, teal, and other species were found occasionally and seasonally. Table 3.1 shows estimated monthly bird count for two years. Species observed were:

Geese and swans;

- Canada goose (*Branta canadensis*)
- Graylag goose (*Anser anser* (Linnaeus))
- Mute swan (*Cygnus olor* (Gmelin))

Dabbling ducks;

- Mallard (*Anas platyrhynchos*)
- Shoveler (*Anas clypeata* (Linnaeus))
- Pintail (*Anas acuta* (Linnaeus))

Coot;

- Coot (*Fulica atra* (Linnaeus))
- Moorhen (*Gallinula chloropus* (Linnaeus))

Small ducks;

- Tufted ducks (*Aythya fuligula* (Linnaeus))
- Teal (*Anas crecca* (Linnaeus))
- Wigeon (*Anas penelope* (Linnaeus))

Gulls;

- Black headed gulls (*Larus ridibundus* (Linnaeus))

Grebes;

- Little grebes (*Tachybaptus ruficollis* (Pallas))

The greatest numbers of birds residing at the site were dabbling ducks while the least were grebes (*Figure 3.9*). The proportions of coot, small ducks, geese and gulls were nearly equal. I found a large flock of Canada geese only in winter 2004 and numbers of them in the pool were relatively lower in 2005 and 2006.

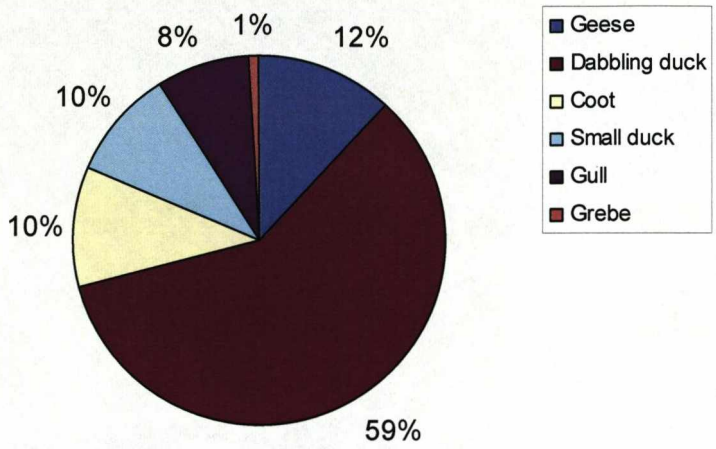


Figure 3.9 Percentage of waterfowl at Brown Moss

Table 3.1 Estimated monthly bird count (bird-day per pool per month) in pool 6

Month	Geese	Dabbling ducks	Coot	Small ducks	Gulls	Grebes	Waterfowl
Nov 04	4,800	3,540	240		780		9,360
Dec	1,450	3,850	284	186	2,030	62	7,860
Jan	128	2,820	314	124	924	62	4,370
Feb	108	2,800	392	140	548		3,990
Mar	274	1,630	323	124	118		2,470
Apr	88	860	122	120	88		1,280
May	125	706	177	62			1,070
Jun		1,220	320		30		1,570
Jul		1,094	250				1,340
Aug		530	240				770
Sep	180	330					510
Oct		1,130		93			1,220
Nov 05	120	2,980		78			3,180
Dec		2,890		548	31		3,470
Jan		3,740		1,690	190	217	5,840
Feb		2,340	90	1,700	56		4,260
Mar	186	1,508	260	1,870		31	3,860
Apr		468	228				670
May		672	286	93			1,050
Jun	60	876	424	30			1,390
Jul	104	1,500	1,150	93	217	124	3,180
Aug	182	2,230	1,120			62	3,600
Sep	1,170	3,002	1,038	60	540	30	5,840
Oct 06		1,420	550	310	465		2,750
Total*	**8,960**	**44,100**	**7,800**	**7,390**	**6,020**	**588**	**74,900**

* **unit** : bird-day per pool per two years

More species and higher numbers of wildfowl were found in winter and fewer in summer. Numbers of waterfowl decreased gradually toward summer (*Figure 3.10*). The densities of waterfowl reached a peak (108 ind. ha^{-1} day^{-1}) in November 2004 when large numbers of Canada geese were roosting on the pool and the lowest densities were 6 ind. ha^{-1} day^{-1} in September 2005.

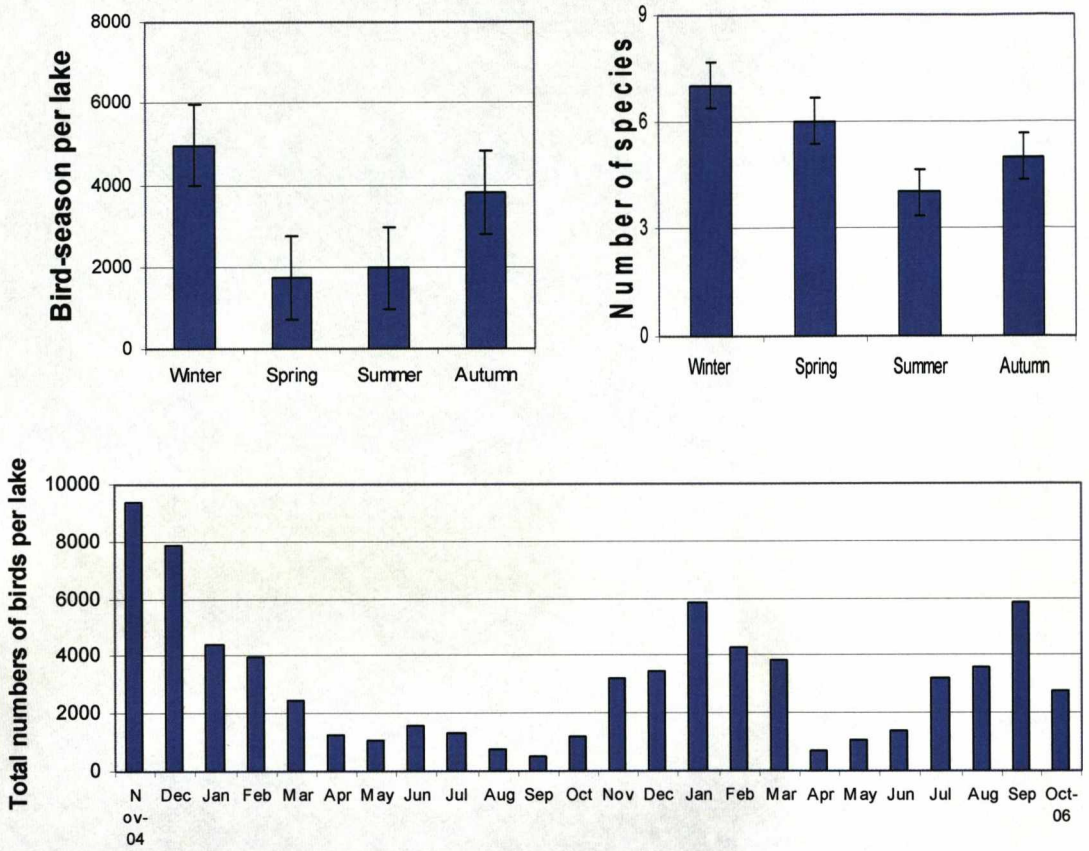


Figure 3.10. Average seasonal numbers and species of waterfowl (above) and variation of numbers of waterfowl in Brown Moss from November 04 – October 06 (below) (n = 2)

I did the diurnal counts every hour in 2004-5 to investigate numbers of birds present in the pool throughout the day and discovered that most water birds were consistently present from morning to evening (*Figure 3.11*). The presence of birds throughout the day did not change much in summer when birds were not abundant. I did not study and count birds at night but night surveys elsewhere with infrared optics have revealed similar bird numbers to those by day in the study area (Schmieder et al. 2006).

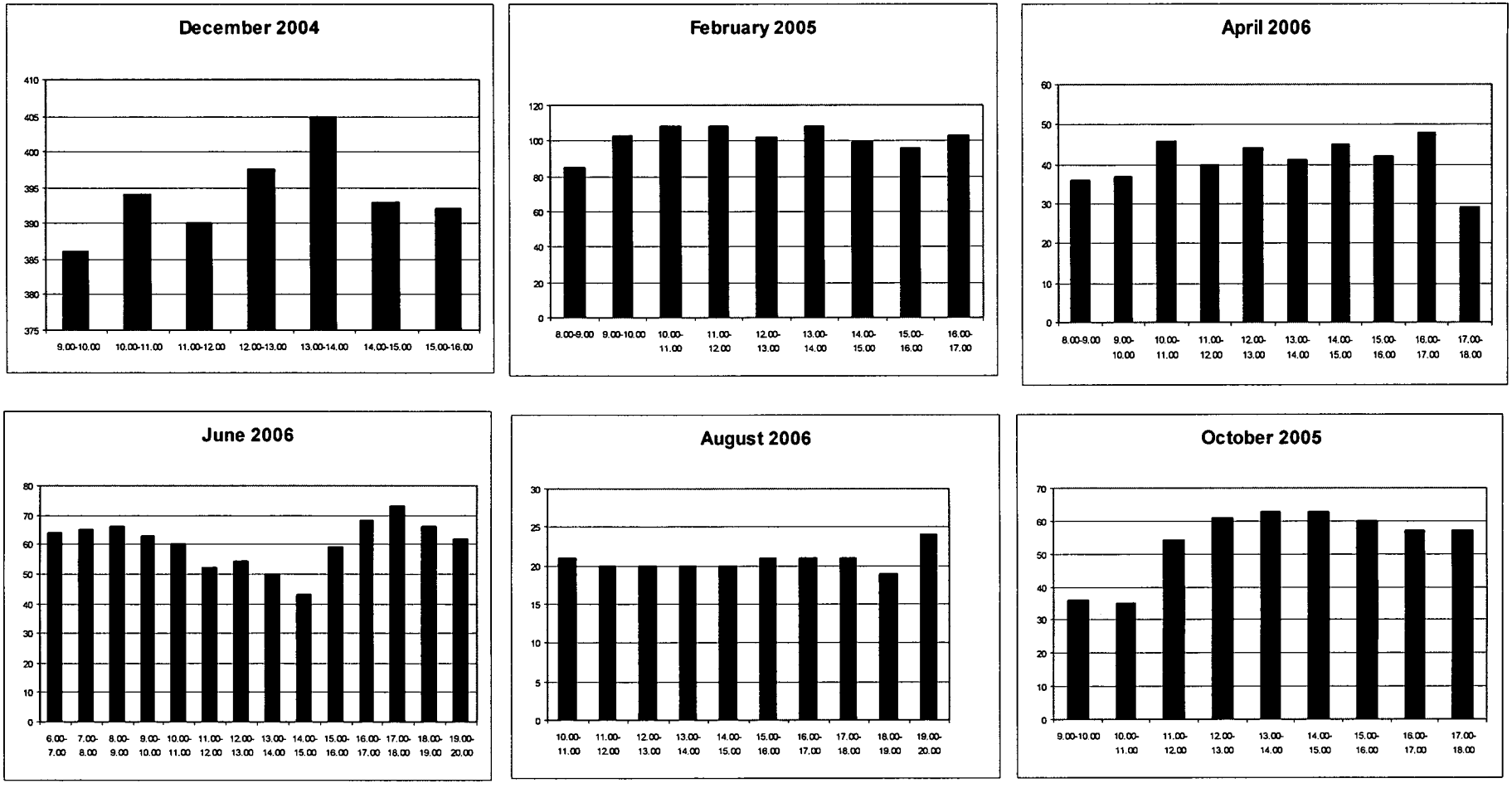


Figure 3.11 Total hourly bird count during daylight at Brown Moss (unit : birds per hour)

In the first few months of data collection, a large flock of Canada geese congregated densely in the pool and strong significant correlations ($r = 0.871$, $P = 0.01$, n = 11) were found between waterfowl number and ammonium nitrogen. Figure 3.12 shows concentrations of ammonia increasing in November 2004, which was consistent with numbers of Canada geese, gathering in the pool. I also looked for correlation between geese and other nutrients but found none.

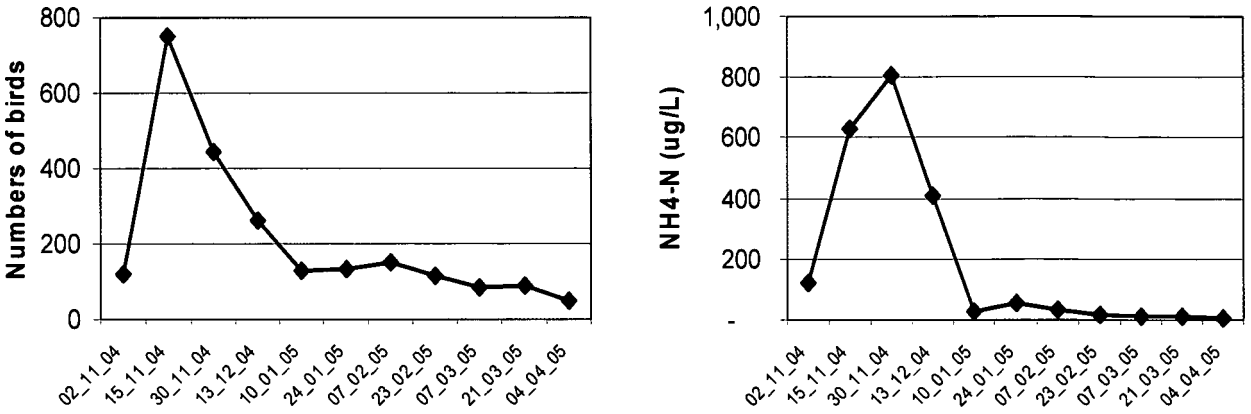


Figure 3.12 Numbers of Canada geese and variations of ammonium in pool 6 (data from bird counts from November 04 to April 05)

Daily nutrient loading rates produced by different groups of birds were obtained from literature reviews (*Table 3.2*). I used daily loading rates from Table 3.2 and monthly bird counts obtained from our survey (*Table 3.1*) to estimate monthly TN and TP loading rates (*Tables 3.4 and 3.5*). Monthly nutrient loading rates by dabbling ducks are shown in Table 3 as an example of the calculation.

Table 3.2 Daily nutrient loading rates for various bird groups

Type of birds	Average Body weight (kg)	Total Nitrogen (mg dry weight per day)	Total Phosphorus (mg dry weight per day)
Geese	2.56[1]	1,570[1]	490[1]
Dabbling ducks	1.18[1]	562[2]	178[2]
Coots	0.54[4]	251[2]	79[2]
Small ducks	0.47[4]	200[2]	69[2]
Small gulls	0.26[6]	122[2]	38[5]
Grebes	0.17[2]	71[2]	22[2]

Remark; (Manny et al. 1994)[1], (based on goose and gull calculation)[2], (Fredrickson, 1969)[3], (Don and Donovan, 2002)[4], (Gould and Fletcher, 1978[5] in Moss and Leah, 1982) and [6] based on estimation

Table 3.3 Estimation of monthly TN and TP produced by dabbling ducks

Month	Estimated dabbling duck-day per month (duck-days)	TN produced by dabbling ducks (mg d.w. $bird^{-1}day^{-1}$	TP produced by dabbling ducks (mg d.w $bird^{-1}day^{-1}$	Monthly TN loading rate (kg $month^{-1}pool^{-1}$	Monthly TP loading rate (kg $month^{-1}pool^{-1}$
Nov 04	3,540	562	178	1.99	0.63
Dec	3,850	562	178	2.16	0.69
Jan	2,820	562	178	1.59	0.50
Feb	2,800	562	178	1.58	0.50
Mar	1,630	562	178	0.91	0.29
Apr	860	562	178	0.48	0.15
May	706	562	178	0.4	0.13
Jun	1,220	562	178	0.69	0.22
Jul	1,094	562	178	0.61	0.19
Aug	530	562	178	0.3	0.09
Sep	330	562	178	0.19	0.06
Oct	1,130	562	178	0.63	0.20
Nov	2,980	562	178	1.67	0.53
Dec	2,890	562	178	1.62	0.51
Jan	3,740	562	178	2.1	0.67
Feb	2,340	562	178	1.32	0.42
Mar	1,508	562	178	0.85	0.27
Apr	468	562	178	0.26	0.08
May	672	562	178	0.38	0.12
Jun	876	562	178	0.49	0.16
Jul	1,500	562	178	0.84	0.27
Aug	2,230	562	178	1.25	0.40
Sep	3,002	562	178	1.69	0.53
Oct-06	1,420	562	178	0.80	0.25

Table 3.4 Estimated TN produced by birds (kg pool^{-1} month^{-1})

Month	Geese	Dabbling ducks	Coot	Small ducks	Gull	Grebe	Total (kg month^{-1})
Nov 04	7.53	1.99	0.060		0.095		9.67
Dec	2.27	2.16	0.071	0.037	0.247	0.004	4.80
Jan	0.20	1.59	0.079	0.025	0.113	0.004	2.01
Feb	0.17	1.58	0.098	0.028	0.067		1.94
Mar	0.43	0.91	0.081	0.025	0.014		1.46
Apr	0.14	0.48	0.031	0.024	0.011		0.69
May	0.20	0.40	0.044	0.012			0.65
Jun		0.69	0.080		0.004		0.77
Jul		0.61	0.063				0.68
Aug		0.30	0.060				0.36
Sep	0.28	0.19					0.47
Oct		0.63		0.019			0.65
Nov	0.19	1.67		0.016			1.88
Dec		1.62		0.110	0.004		1.74
Jan		2.10		0.338	0.023	0.015	2.48
Feb		1.32	0.023	0.354	0.007		1.70
Mar	0.29	0.85	0.065	0.375		0.002	1.58
Apr		0.26	0.057				0.32
May		0.38	0.072	0.019			0.47
Jun	0.09	0.49	0.106	0.006			0.70
Jul	0.16	0.84	0.287	0.019	0.026	0.009	1.35
Aug	0.29	1.25	0.281			0.004	1.83
Sep	1.83	1.69	0.261	0.012	0.066	0.002	3.86
Oct-06		0.80	0.138	0.062	0.057		1.06
						Average	**1.8**

Table 3.5 Estimated TP produced by birds (kg $pool^{-1}$ $month^{-1}$)

Month	Geese	Dabbling ducks	Coot	Small ducks	Gull	Grebe	Total (kg $month^{-1}$)
Nov 04	7.53	1.99	0.060		0.095		3.03
Dec	2.27	2.16	0.071	0.037	0.247	0.004	1.51
Jan	0.20	1.59	0.079	0.025	0.113	0.004	0.63
Feb	0.17	1.58	0.098	0.028	0.067		0.61
Mar	0.43	0.91	0.081	0.025	0.014		0.46
Apr	0.14	0.48	0.031	0.024	0.011		0.22
May	0.20	0.40	0.044	0.012			0.21
Jun		0.69	0.080		0.004		0.24
Jul		0.61	0.063				0.21
Aug		0.30	0.060				0.11
Sep	0.28	0.19					0.15
Oct		0.63		0.019			0.21
Nov	0.19	1.67		0.016			0.59
Dec		1.62		0.110	0.004		0.55
Jan		2.10		0.338	0.023	0.015	0.79
Feb		1.32	0.023	0.354	0.007		0.55
Mar	0.29	0.85	0.065	0.375		0.002	0.51
Apr		0.26	0.057				0.10
May		0.38	0.072	0.019			0.15
Jun	0.09	0.49	0.106	0.006			0.22
Jul	0.16	0.84	0.287	0.019	0.026	0.009	0.43
Aug	0.29	1.25	0.281			0.004	0.58
Sep	1.83	1.69	0.261	0.012	0.066	0.002	1.21
Oct-06		0.80	0.138	0.062	0.057		0.34
						Average	**0.57**

Monthly mean amounts of TN and TP loading by aquatic birds were 1.8 and 0.57 kg per pool, respectively (*Table 3.4, 3.5*). Higher TN and TP loads were found in winter and autumn and lower values in spring and summer. TN and TP derived from the calculation were highest in November and still remained high during winter when waterfowl were most abundant (*Figures 3.13 and 3.14*). After winter, TP and TN contribution decreased because there was only a small group of mallard remaining on the pool. Over all, estimated nutrient input from birds was not statistically correlated with concentrations of nutrients in the pool. However, significant correlation between numbers of bird and increased ammonium concentrations was found in winter 2004-5 (*Fig. 3.12*).

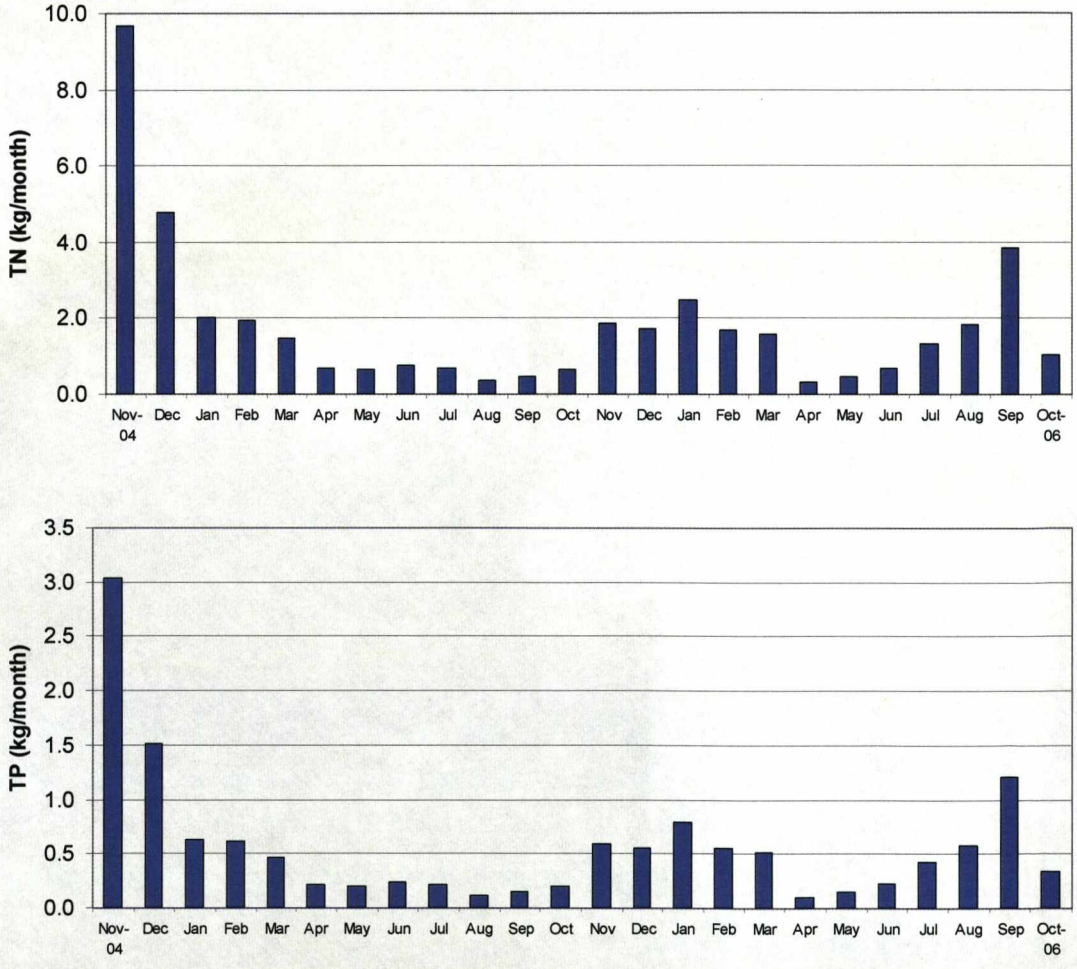


Figure 3.13 Estimated monthly loading of total nitrogen and phosphorus (kg pool^{-1} month^{-1}) (n = 2) by waterfowl from 2004-2006 to pool 6

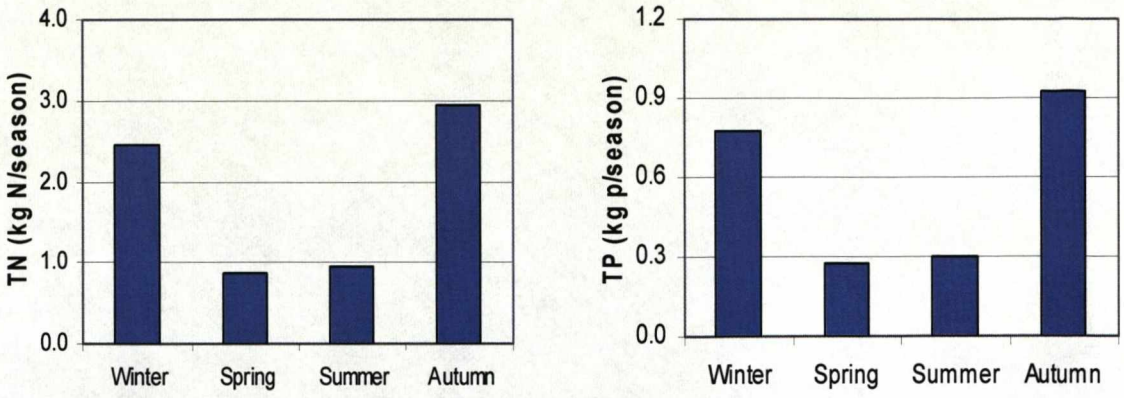


Figure 3.14 Seasonal loading of nutrients by waterfowl in pool 6, Brown Moss

To estimate loading rates of TN and TP by birds to pool 6, I divided monthly loading rates by area of the pool:

$$= \frac{\text{Monthly numbers of waterfowl (group separated)} \times \text{excretion rates (mg yr}^{-1})}{\text{Area of pool (m}^2)}$$

= The amount of TP introduced by birds (mg m^{-2} yr^{-1})

$$= \frac{6{,}800{,}000}{29{,}000}$$

= 234 mg m^{-2} yr^{-1}

= The amount of TN introduced by birds (mg m^{-2} yr^{-1})

$$= \frac{21{,}500{,}000}{29{,}000}$$

= 741 mg m^{-2} yr^{-1}

Therefore, estimated TP and TN loadings by waterfowl to pool 6 at Brown Moss were 234 and 741 mg m^{-2} yr^{-1}.

3.3.2 Ground water

Levels of ground water fluctuated dramatically in the boreholes. The lowest and highest levels recorded were zero and 72 cm (measured from the bottom of the borehole to the groundwater surface) in September 2005 and in May 2006, respectively. Figure 13 shows that water in the boreholes declined in summer 2005 and completely dried out in September. Changes in ground water table in boreholes were also consistent with that in pool 6 and a significant positive correlation between water levels in pool 6 and in boreholes was found ($r = 0.529$, $P = 0.016$) (*Figure 3.15*). About 72% of water may enter the pool directly from rain or is absorbed by tress and soil particles and never reach the pool.

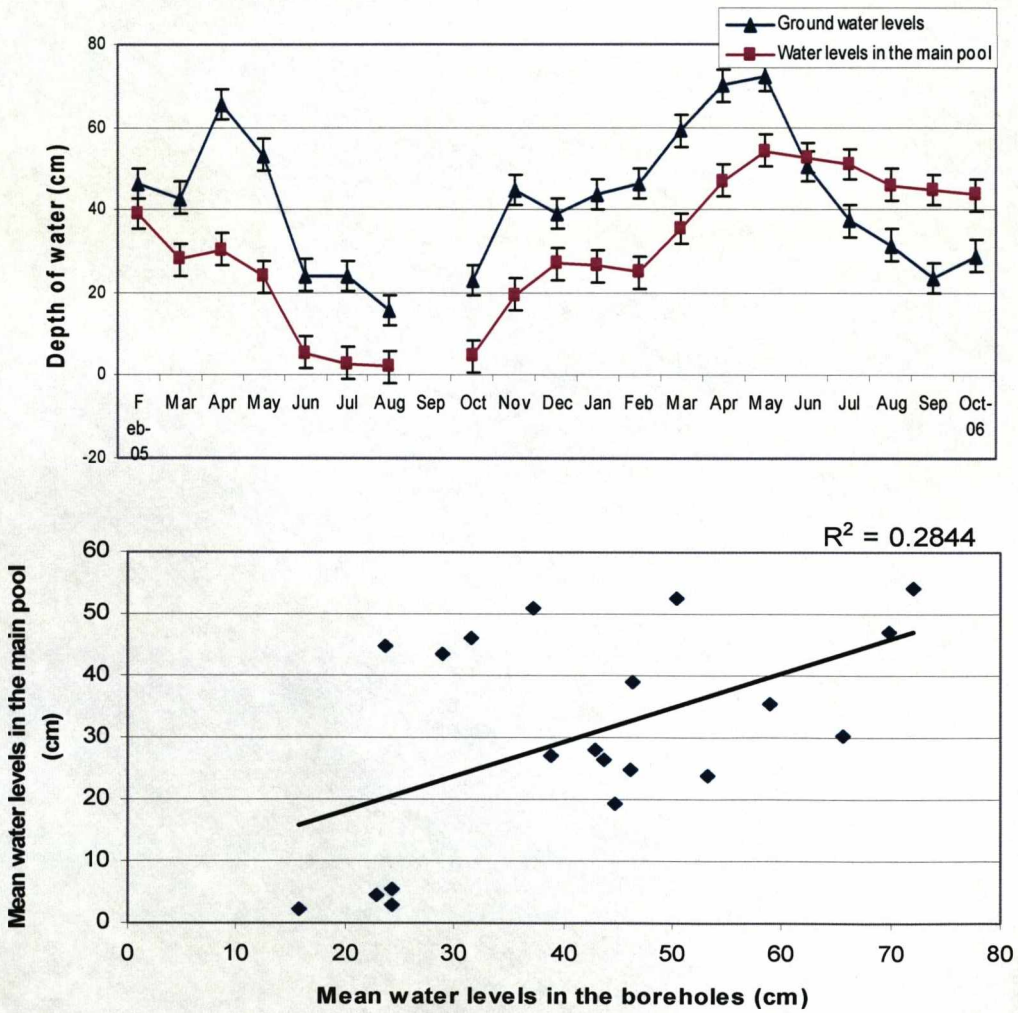


Figure 3.15 Relationship between water levels in boreholes and in pool 6

Changes in water chemistry were both positively and negatively related with changes of water levels in boreholes. Figure 3.16 shows that concentrations of nutrients increased on occasions when water levels decreased and increased. For example, average concentrations of NO_3^--N and TN from boreholes were high when ground water levels decreased in June and July 2005 and this may be the result of evaporative concentration. Concentrations of SRP, TP, NO_3^--N, NH_4^+-N and TN also tended to increase in spring and early summer when ground water level increased and this indicated contaminated water transported by surface and ground water into the boreholes. Afterwards, concentrations of all parameters measured started to decline throughout summer and increased again in autumn. NH_4^+-N was significantly correlated with water levels but other variables were not (*Table 3.6*). Concentration of NH_4^+-N was also significantly correlated with SRP and TP.

Table 3.6 Correlation coefficients of changes of water levels and nutrients in the boreholes (n = 43)

Variables	**Depth**	**SRP**	**TP**	**NO_3^--N**	**NH_4^+-N**
SRP	0.260				
TP	0.265	-			
NO_3^--N	0.014	-0.256	-0.294		
NH_4^+-N	0.583**	0.776**	0.810**	-0.051	
TN	0.052	-0.135	-0.166	-	-

Remark; **. Correlation is significant at the 0.01 level and

- Auto correlated

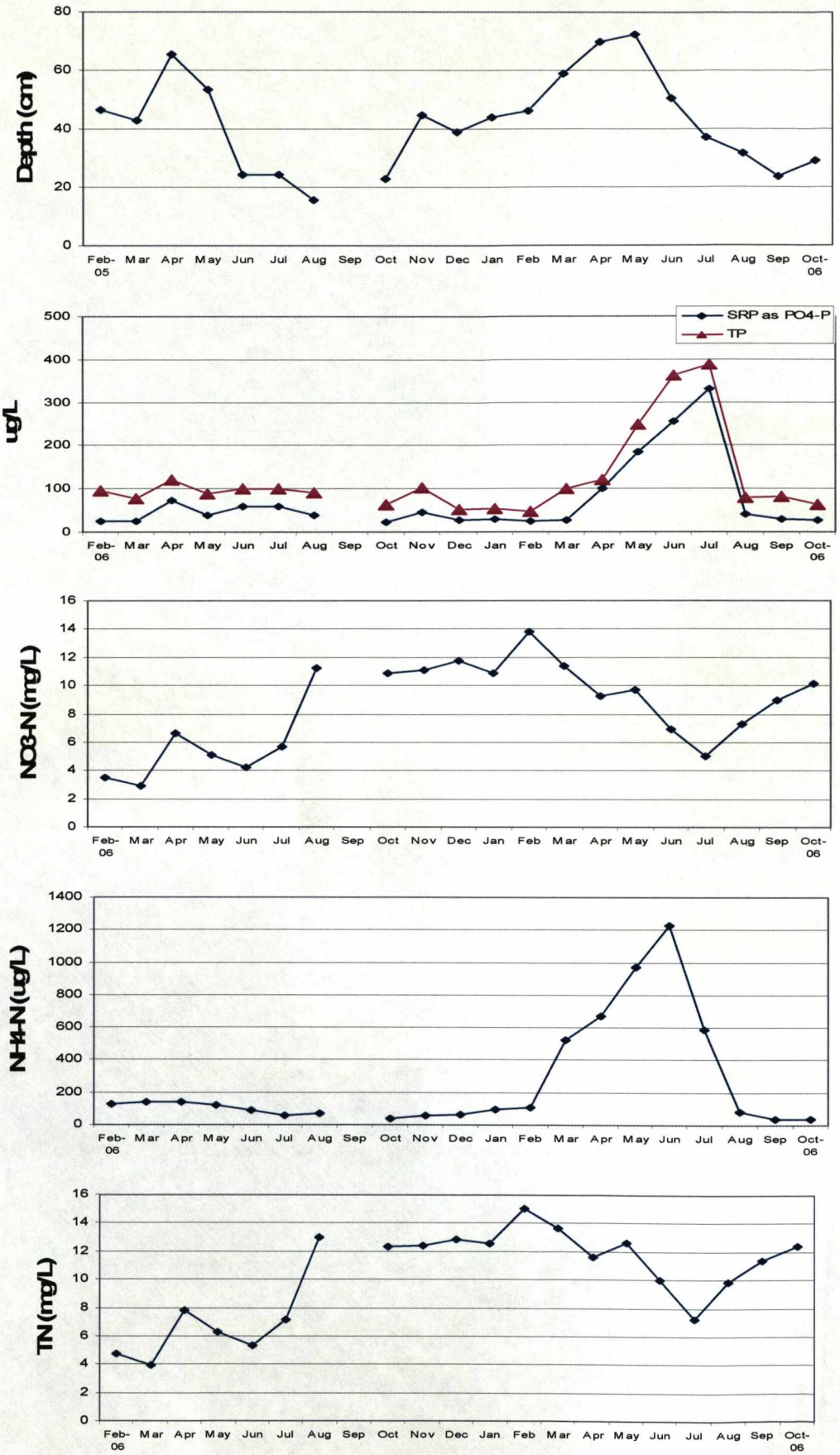


Figure 3.16 Changes of water chemistry and water levels in boreholes

The highest amounts of SRP, TP, NH_4^+-N, NO_3^--N and TN were detected in borehole 5 and the highest quantity of NO_3^--N was found in borehole 7 (*Table 3.7*). The lowest amounts of SRP, NO_3^--N and TN were in borehole 6 and the lowest quantities of TP, NH_4^+-N were in borehole 4 (*Table 3.7*). In borehole 5, NH_4^+-N increased considerably at the same time when water level in the borehole increased and a significant correlation between water level and the amount of NH_4^+-N was found ($r = 0.745$, $P = 0.013$).

Table 3.7 Mean concentration with standard deviation (n = 43) of plant nutrients in boreholes around the site

Boreholes	SRP ($\mu g\ L^{-1}$)	TP ($\mu g\ L^{-1}$)	NO_3^--N ($mg\ L^{-1}$)	NH_4^+-N ($\mu g\ L^{-1}$)	TN ($mg\ L^{-1}$)
1	67.6±62.5	147±94	8.3±7.5	86±67.8	10.5±7.7
2	46.3±31.8	70.6±41.5	1.8±2.7	59.6±77.3	2.4±3.2
3	235±360.4	293±440	2.7±2.1	543±795.2	4.2±2.4
4	12.4±5.2	**37.8±28.9**	1.6±1.4	**45.2±42.4**	2.5±1.5
5	**444±537.9**	**621±610.6**	19.6±15.8	**3008±3438.4**	**25.1±18.14**
6	**9.2±5.5**	39.8±61.3	**0.5±0.3**	82.8±49.8	**0.97±0.3**
7	21.2±14.6	41.9±19	**22.3±5.9**	64±39.5	**25.1±6.1**
8	15.8±12.7	38.9±20.6	6±5.5	165±196.5	7±6.2
9	47.6±23	82.1±29	1.1±1.2	76.4±57	2.6±1.5
10	20.9±17.3	130±144.4	13.9±7	389±191.7	15.8±7
Average	**92±140.6**	**150±183.2**	**7.8±8.1**	**452±913.3**	**9.6±9.3**

Remark : Bold black and **red colour** represents the highest and lowest amounts of nutrients, respectively.

Water chemistry of ground water in each borehole differed, especially between those located on the fringe of Brown Moss and those in the middle of the site. Average TP concentrations showed higher values in boreholes 1, 3, 5, 9 and 10 and similarly TN concentrations were high in boreholes 1, 5, 7 and 10. Lower amounts of nutrients were found in boreholes 2, 4, 6 and 8 installed in the inner areas of Brown Moss nature reserve (*Figure 3.17 and 3.18*).

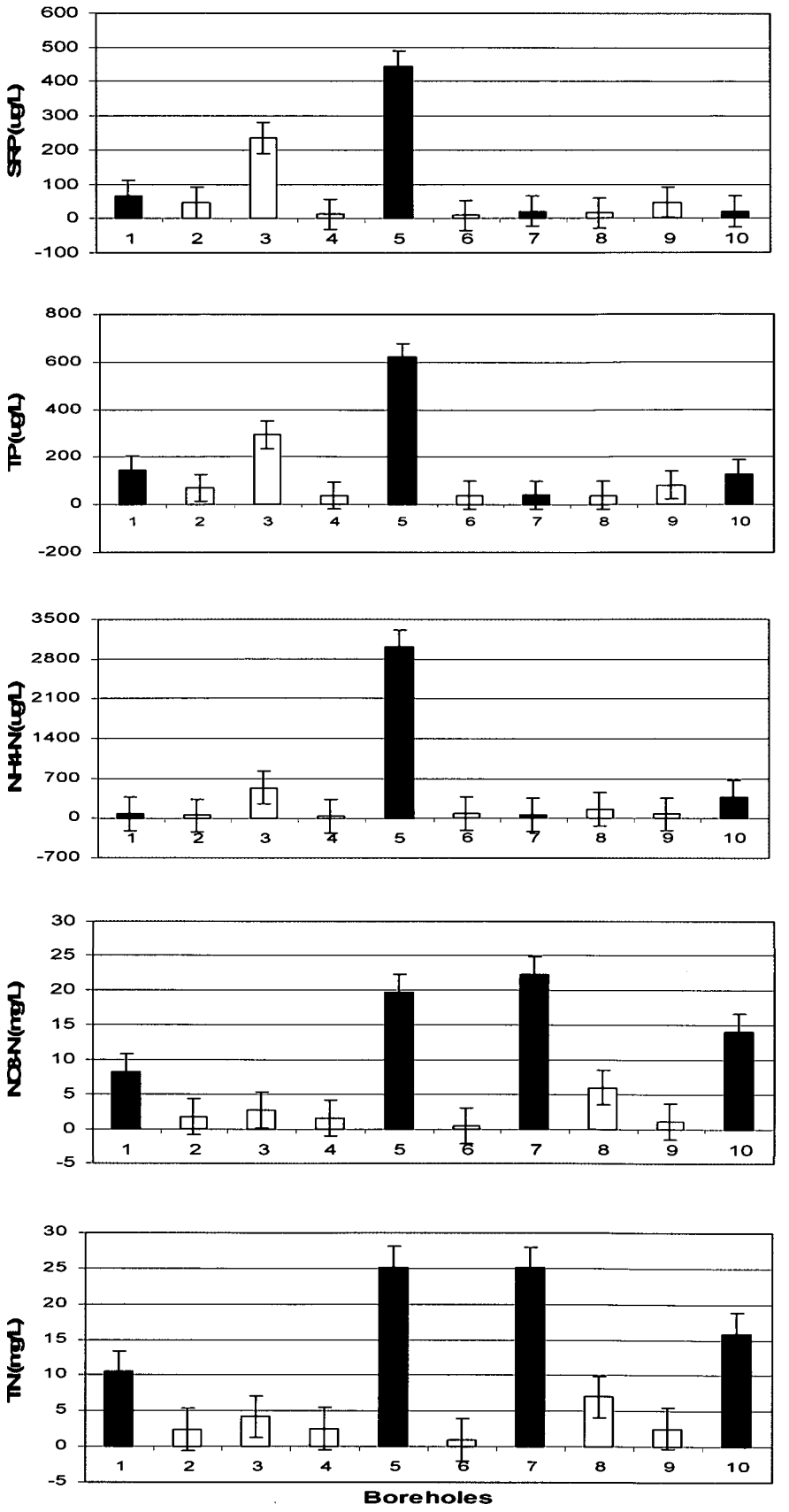


Figure 3.17 Average concentrations of nutrients in boreholes (black columns represent boreholes near the edge of the site and white columns those boreholes in inner areas) with standard error (n = 2)

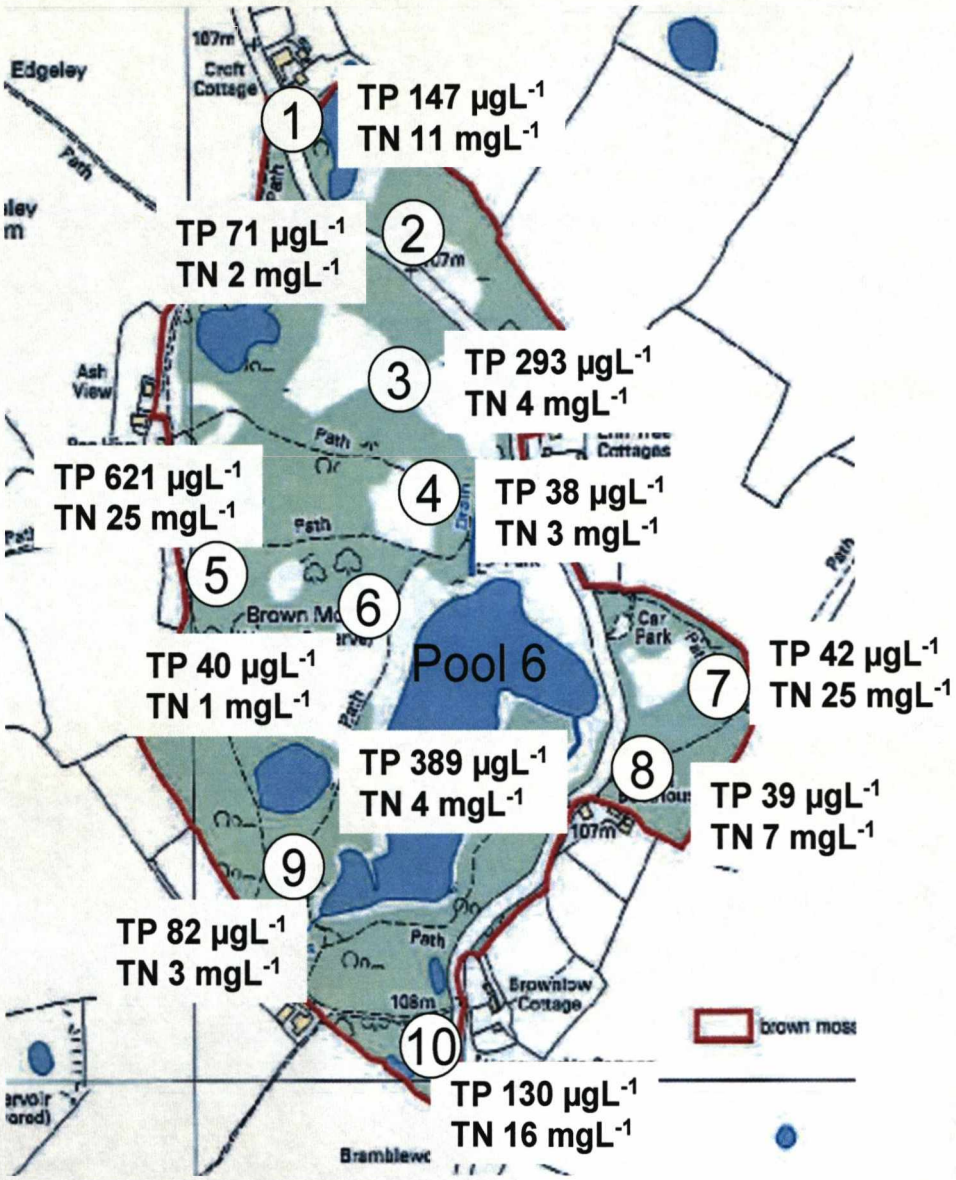


Figure 3.18 Average concentrations of nutrients in boreholes situated around the site

I used Bray-Curtis analysis to determine similarity of water chemistry among boreholes (*Figure 3.19*). Concentrations of nutrients (SRP, TP, NH_4^+-N, NO_3^--N and TN) in borehole 4 were similar to those in boreholes 6, 7 and 8 while water chemistry of boreholes 1, 2 and 9 were categorised in the same group.

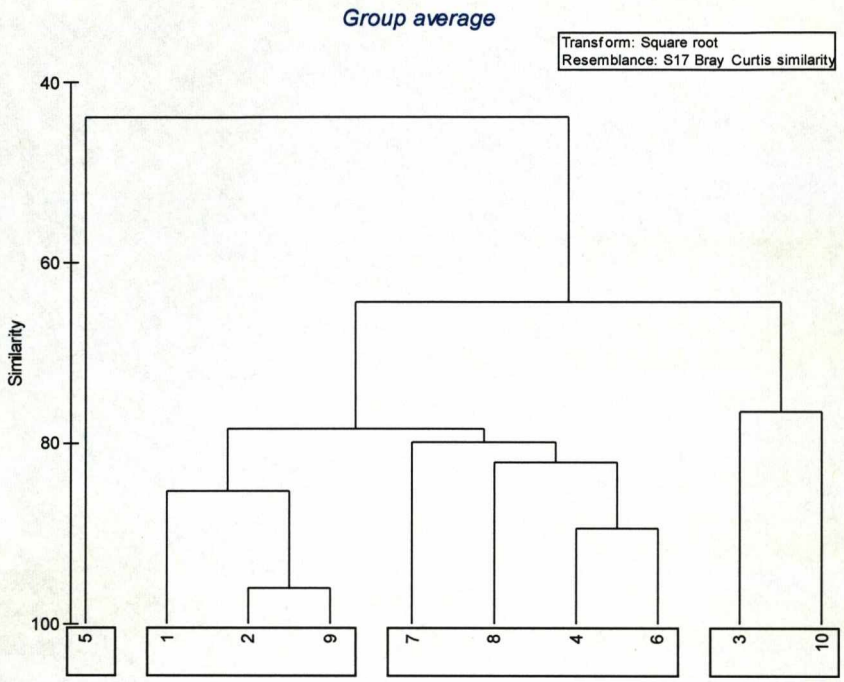


Figure 3.19 Bray-Curtis similarity of nutrients (SRP, TP, NH_4^+-N, NO_3^--N and TN) among boreholes

Similarity between fluctuations of water levels in boreholes and in pool 6 suggested that pool 6 receives water from underground. I also found significant correlation between nitrate in borehole 7 and in pool 6 ($r = 0.487$, $P = 0.035$) (*Table 3.8*). However, nutrients are non-conservative and so such a relationship could be misleading in determining the sources of soil water to the pool.

Table 3.8 Correlation coefficients between concentrations of nitrate in the boreholes and in pool 6 (n = 43)

Variable	NO_3^--N_{pool6}	NO_3^--N_{B1}	NO_3^--N_{B2}	NO_3^--N_{B3}	NO_3^--N_{B4}	NO_3^--N_{B5}	NO_3^--N_{B6}	NO_3^--N_{B7}	NO_3^--N_{B8}	NO_3^--N_{B9}
NO_3^--N_{B1}	0.448*									
NO_3^--N_{B2}	-0.282	-.061								
NO_3^--N_{B3}	-0.085	0.583*	0.209							
NO_3^--N_{B4}	0.451	0.705**	0.182	0.262						
NO_3^--N_{B5}	-0.534	0.941**	0.391	0.528	0.630					
NO_3^--N_{B6}	-0.135	0.569	0.308	0.419	0.300	0.475				
NO_3^--N_{B7}	0.487*	0.823**	-0.150	0.513	0.733**	0.816**	0.245			
NO_3^--N_{B8}	-0.163	-0.100	0.149	-0.283	0.456	-0.020	-0.018	-0.074		
NO_3^--N_{B9}	-0.276	0.056	0.339	0.112	0.410	0.447	0.335	-0.082	0.630*	
NO_3^--N_{B10}	0.211	0.913**	0.140	0.725**	0.401	0.830*	0.597	0.740*	-0.410	-0.127

Remark; **.Correlation is significant at the 0.01 level and

*. Correlation is significant at the 0.05 level.

B stands for borehole

As described in section 3.2.2, estimation of nutrient loading from ground water was calculated based on volume of water entering pool 6 (runoff), mean concentrations of plant nutrients from specific boreholes and area of catchment of pool 6. By considering Bray-Curtis analysis using values of alkalinity and conductivity, and concentrations of winter TN which is assumed to be conservative, I assumed that pool 6 receives most water from area where boreholes 1, 2, 3, 4 and 8 were situated (see *Figures 3.20 and 3.21*). I did not include the areas where boreholes 5, 6, 7, 9 and 10 were located because the results showed that they were less similar in conservative ions than those mentioned. Furthermore, although there is a spring around borehole 9 that supplies water to a small pond and then to pool 6, the small pond is not connected to pool 6 throughout the year especially in summer and accordingly, I excluded borehole 9.

Thus, the possible catchment area of pool 6 around boreholes 1, 2, 3, 4 and 8 was calculated as 120,000 m^2. Mean TP and TN values from boreholes 1, 2, 3, 4 and 8 were 117 and 5,320 mg m^{-3}, respectively (*Figure 3.20*).

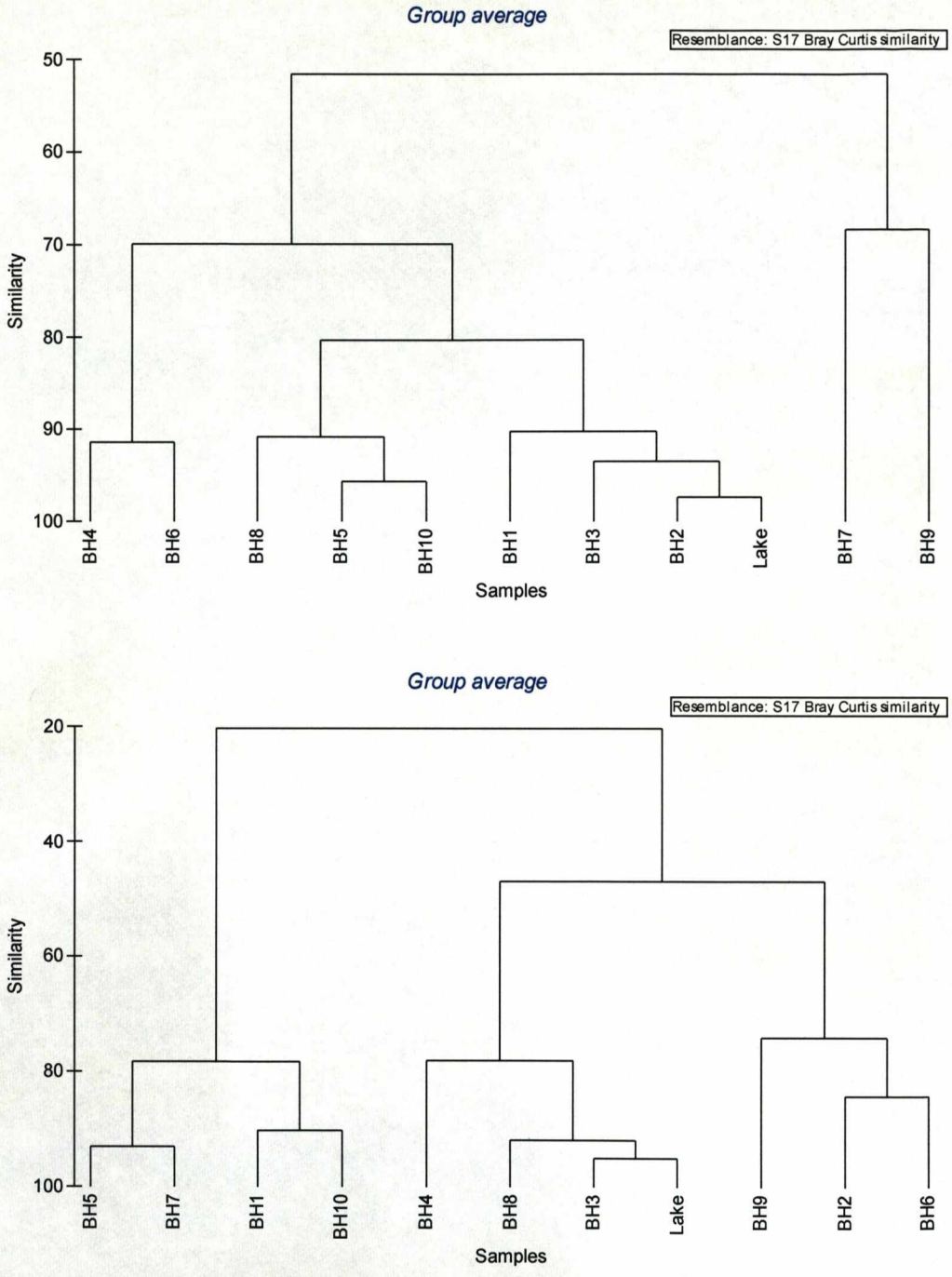


Figure 3.20 Cluster analysis of alkalinity and conductivity (above), and winter TN (below) among boreholes and the pool

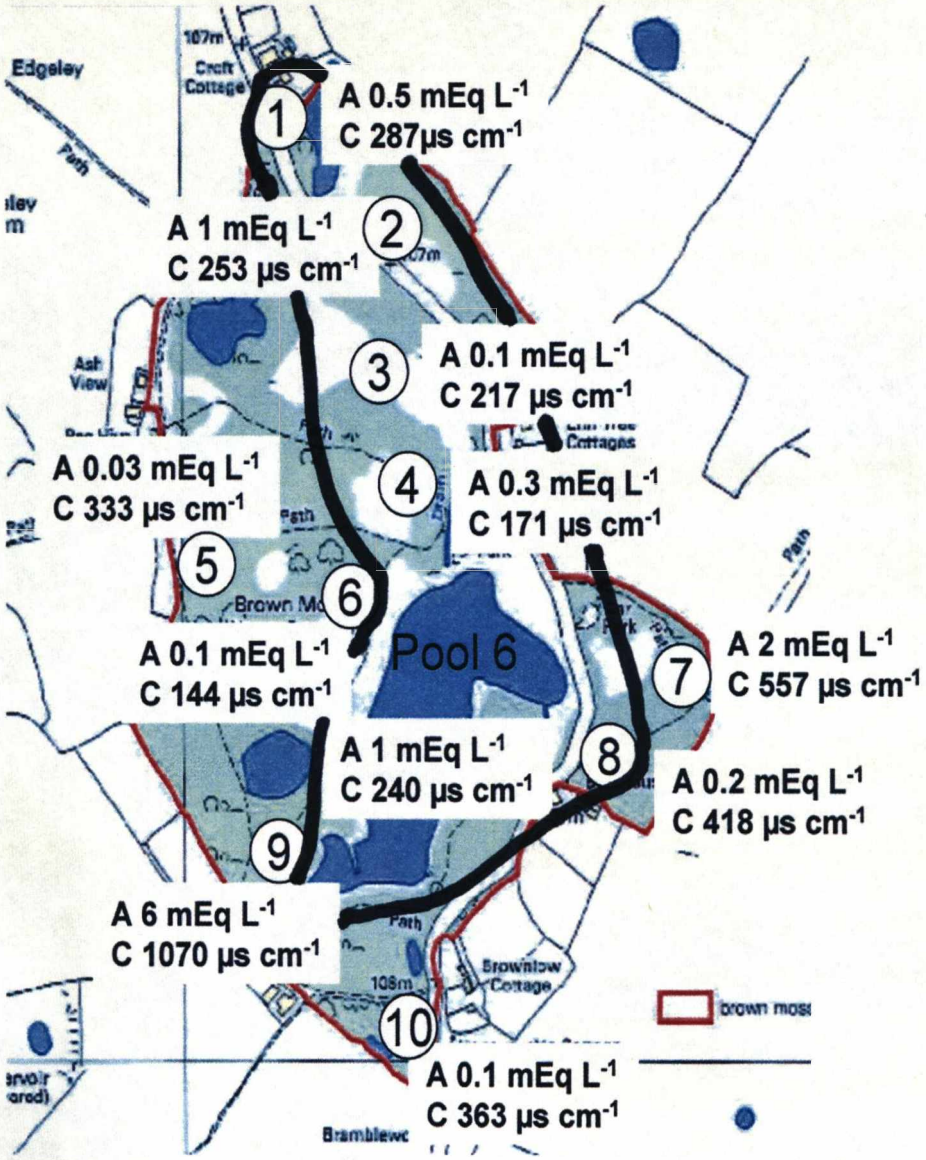


Figure 3.21 The possible catchment area of pool 6 (black thick line) and alkalinity (A) and conductivity (C) of groundwater in the boreholes measured around Brown Moss

I calculated runoff in the catchment area by considering annual rainfall (0.7 m yr^{-1}) and evaporation (0.57 m yr^{-1}):

Runoff = rain fall – evaporation = 0.7 – 0.57 = 0.13 m yr^{-1}

Amounts of TP and TN entry are equal to

$$\frac{\text{Average TP and TN concentration from specific boreholes (mg } m^{-3}) \times \text{(precipitation - evaporation) (m } yr^{-1}) \times \text{catchment area of pool 6(}m^2)}{\text{Area of pool 6(}m^2)}$$

Amount of TP entry = $\frac{117 \times 0.13 \times 120{,}000}{29{,}000}$ mg m^{-2} yr^{-1}

Amount of TP entry = 63 mg m^{-2} yr^{-1}

Amount of TN entry = $\frac{5{,}320 \times 0.13 \times 120{,}000}{29{,}000}$ mg m^{-2} yr^{-1}

Amount of TN entry = 2,860 mg m^{-2} yr^{-1}

Therefore, amounts of TP and TN entry to the pool from ground and soil water were 63 and 2,860 mg m^{-2} yr^{-1}, respectively.

3.3.3 Rain water

More rain fell in winter and less in summer in 2006. Figure 3.22 illustrates fluctuations of water chemistry in rain water. Nutrients in rain water were high in winter and decreased gradually toward spring. Afterwards, nutrients increased again and remained high throughout summer until October when quantities of nutrients declined. pH of rain water measured at the University of Liverpool was slightly acid with an average of 6.1 and average alkalinity and conductivity were 0.086 mEq L^{-1} and 76.37 μS cm^{-1}, respectively. Average concentrations of SRP, TP, NO_3^--N, NH_4^+-N and TN, followed by SD (n = 19) in rain water were 18.9±12 μg L^{-1}, 33.8±22.8 μg L^{-1}, 0.33±0.2 mg L^{-1}, 768±475.3 μg L^{-1} and 1.2±0.7 mg L^{-1}, respectively.

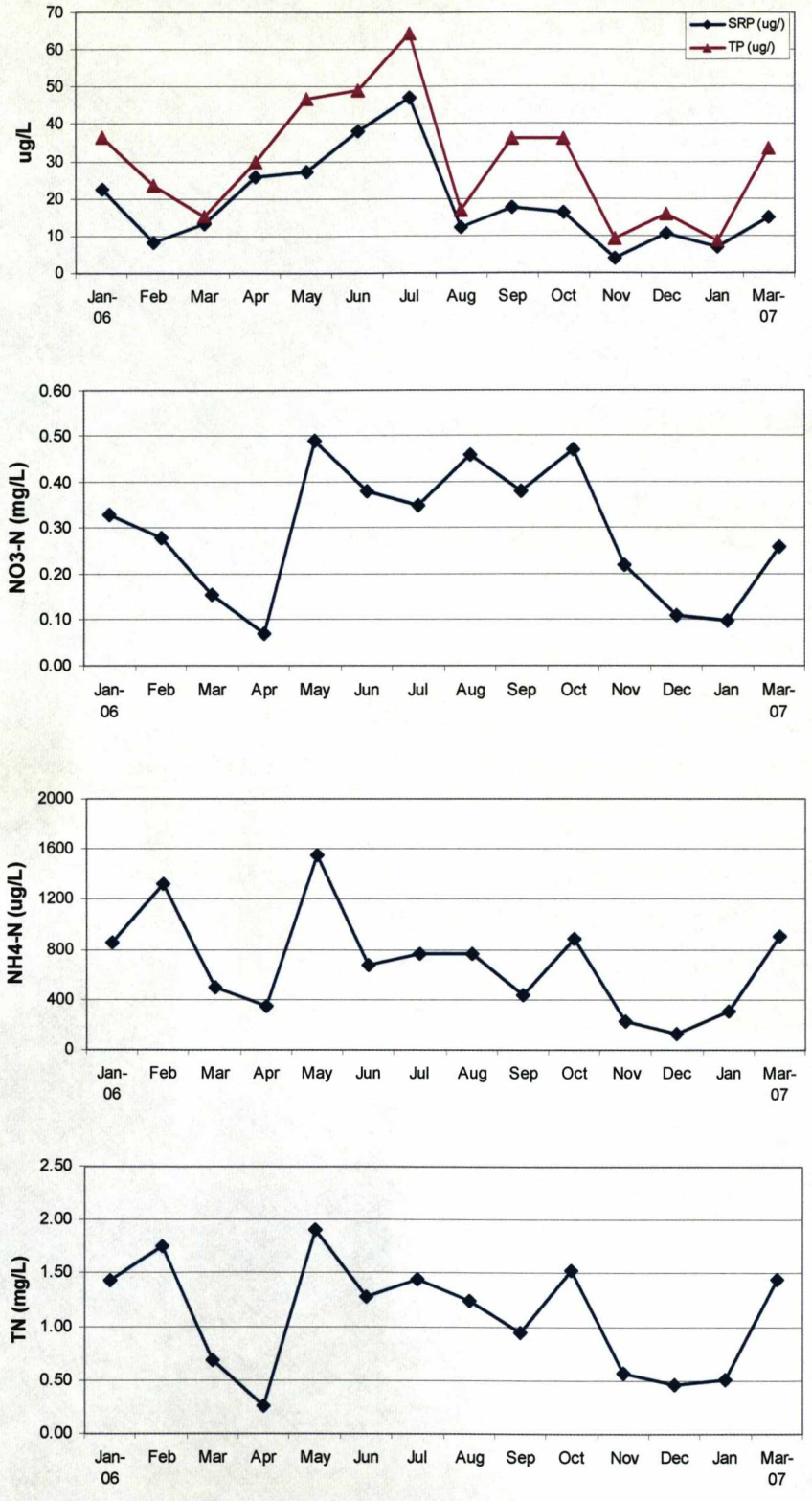


Figure 3.22 Seasonal variation of nutrients in rain water

The amounts of both TP and TN were highest in summer and lowest in winter (*Figure 3.23*). Overall, no statistical differences between average concentrations of TP and TN in each season were observed, although mean values of TP measured in each season were relatively different (F = 1.69, *P = 0.211* and F = 0.53, *P = 0.669*, respectively) *(Figure 3.23)*. Phosphorus was less abundant than nitrogen in rain water. I also applied Pearson analysis to investigate correlation of nutrients between rain water and water in pool 6 and found none, although there were some apparent relationships.

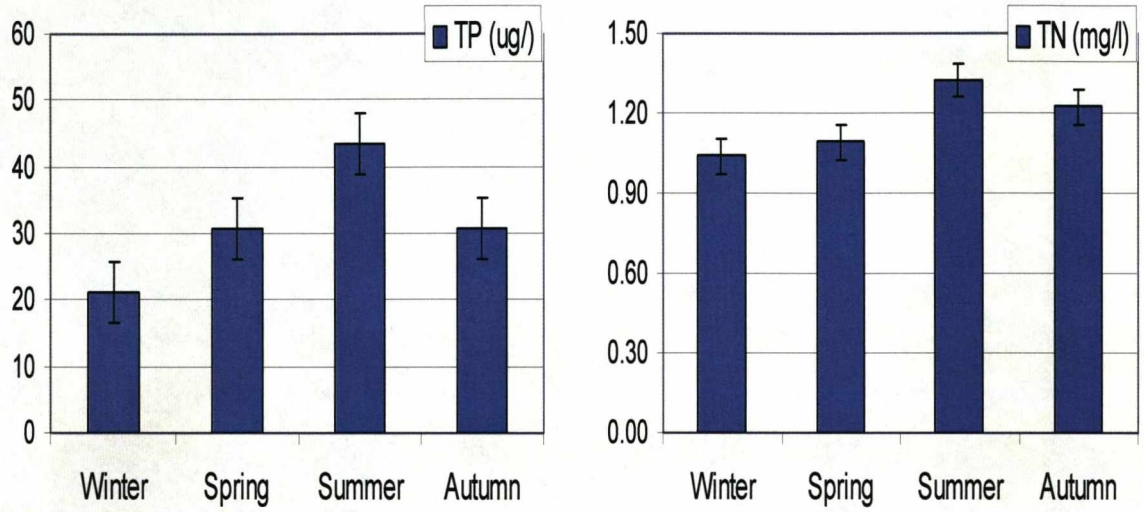


Figure 3.23 Average quantities of TP and TN from rain water in each season with standard error (n = 3)

Table 3.9 Correlation coefficients of variables between rain water and water in pool 6 (n = 10)

Variables	SRP_{rain}	TP_{rain}	NO_3^--N rain	NH_4^+-N rain	TN_{rain}	SRP_{pool}	TP_{pool}	NO_3^--N pool	NH_4^+-N pool
TP_{rain}	-								
NO_3^--N_{rain}	0.060	0.339							
NH_4^+-N_{rain}	-0.103	0.179	0.529						
TN_{rain}	0.039	0.368	0.-	-					
SRP_{pool}	0.556	0.076	0.403	0.205	0.412				
TP_{pool}	-0.434	-0.359	-0.254	0.358	0.276	-			
NO_3^--N_{pool}	-0.234	-0.287	0.204	0.016	0.106	0.256	0.084		
NH_4^+-N_{pool}	-0.003	0.093	0.081	0.144	0.220	0.486	0.248	0.693*	
TN_{pool}	-0.401	-0.314	-0.193	0.305	0.291	0.128	0.955**	-	-

**.= Correlation is significant at the 0.01 level and

*. = Correlation is significant at the 0.05 level.

- = Auto correlated

I estimated nutrient input to pool 6 by using average annual rain fall and concentrations of nutrients as in the equation presented below;

$$= \text{Average annual rainfall (m. yr}^{-1}) \times 1\ \text{m}^2 \times \text{concentration of TP (mg m}^{-3})$$

$$= 0.7 \times 1 \times 33.80$$

$$= 24\ \text{mg m}^{-2}\,\text{yr}^{-1}$$

$$= \text{Average annual rain fall (m. yr}^{-1}) \times 1\ \text{m}^2 \times \text{concentration of TN (mg m}^{-3})$$

$$= 0.7 \times 1 \times 1{,}240$$

$$= 870\ \text{mg m}^{-2}\,\text{yr}^{-1}$$

Therefore, average annual TP and TN loading to pool 6 by rainfall was 24 and 870 mg $m^{-2}\,yr^{-1}$, respectively.

3.3.4 Nutrient release from the sediment

I measured nutrient release from the sediment from three locations in pool 6 at Brown Moss and found that the highest net release of SRP and NH_4^+-N was found at sites A and C (*Figure 3.6*), respectively, while the lowest net SRP and NH_4^+-N release was detected at site B (*Figure 3.24*). Average release rates of total phosphorus and nitrogen (n = 3) from the experiment were 4.57±8.5 and 0.96±30.8 mg m^{-2} day^{-1}, respectively.

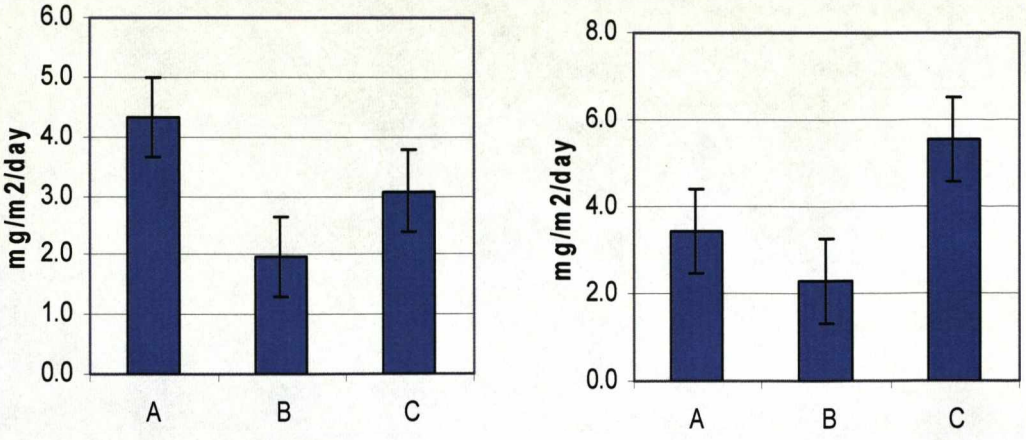


Figure 3.24 Mean values of SRP (left) and NH_4^+-N (right) released from the sediment from three sites over twenty-four hours with standard error (n = 3)

Nutrients released from the sediment at all sites showed similar seasonal trends in that SRP, TP and NH_4^+-N were mostly released in summer (*Figure 3.25*) since the concentrations of SRP, TP and NH_4^+-N in the water column increased over the whole 24-hour incubation period. Winter concentrations of SRP, TP and NH_4^+-N released from the sediment were low in comparison with summer rates of release that were much higher. The release of TN from the sediment increased in the winter months and the release rates of NO_3^--N were negligible or transformed to other N forms in all seasons.

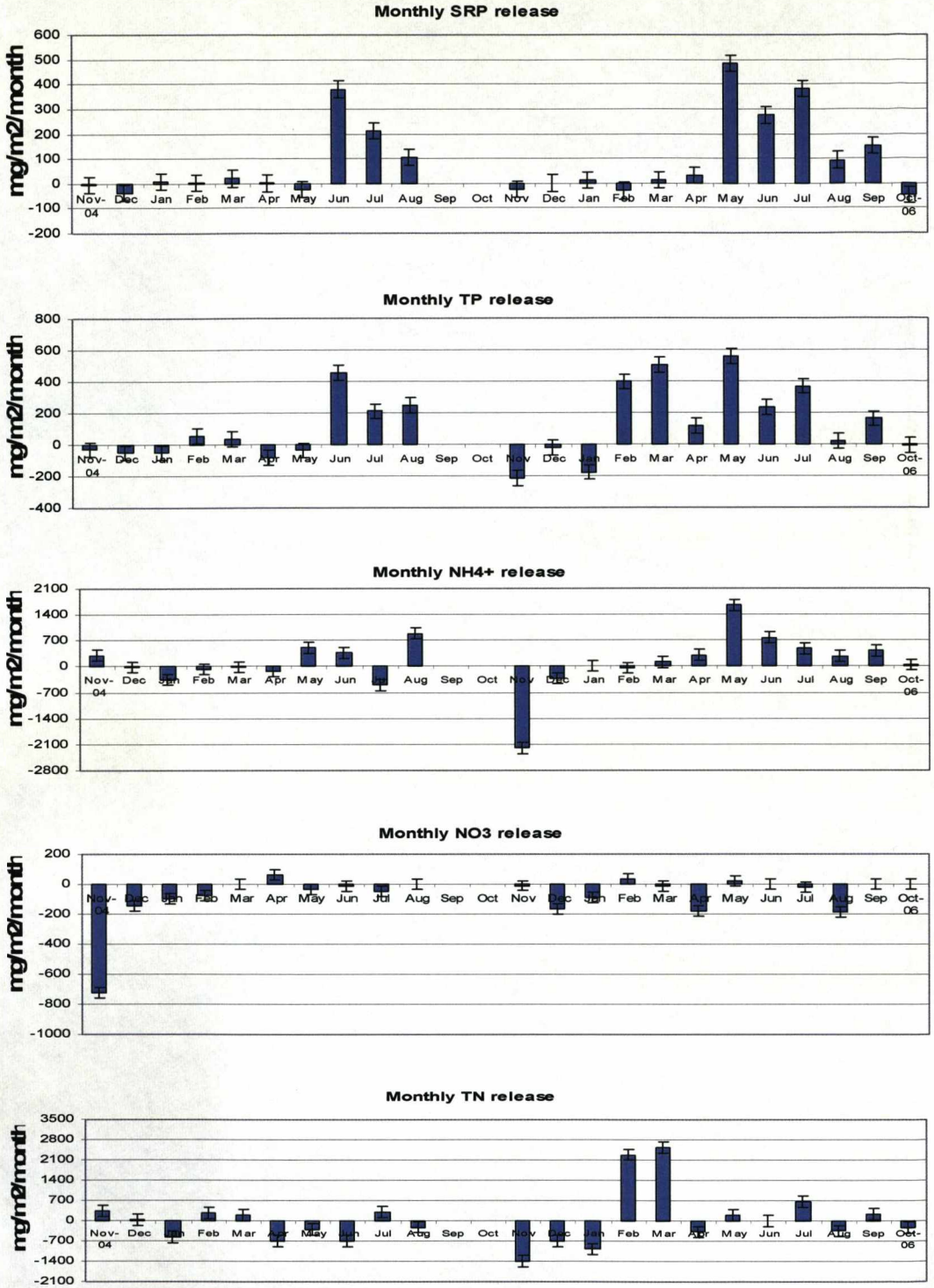


Figure 3.25 Seasonal SRP and ammonium release at Brown Moss over twenty-four months (values above the zero line represent mean release to the water and values below the zero line represent mean uptake by the sediment) with standard error (n = 3).

ANOVA Statistical analysis illustrated that there was no difference in release rates of SRP and NH_4^+-N among sites A, B and C ($F = 0.598$, $P = 0.553$ and $F = 0.062$, $P = 0.940$, respectively) but significant differences in release rates of SRP and NH_4^+-N were found among study months ($F = 5.52$, $P < 0.001$ and $F = 3.213$, $P = 0.002$, respectively).

I estimated nutrient loads to pool 6 from the sediment using mean values of TP and TN release (negative values included) and I then multiplied those values by 365 to give yearly TP and TN release to the pool. Accordingly, yearly TP and TN release to pool 6 at Brown Moss were 1,620 mg m^{-2} yr^{-1} and 340 mg m^{-2} yr^{-1}, respectively.

3.3.5 Water chemistry of pool 6

Water levels in pool 6 at Brown Moss changed considerably in 2005 when water levels had continuously decreased since the winter months (*Figure 3.26*). In September 2005, the whole pool dried up completely for about two months.

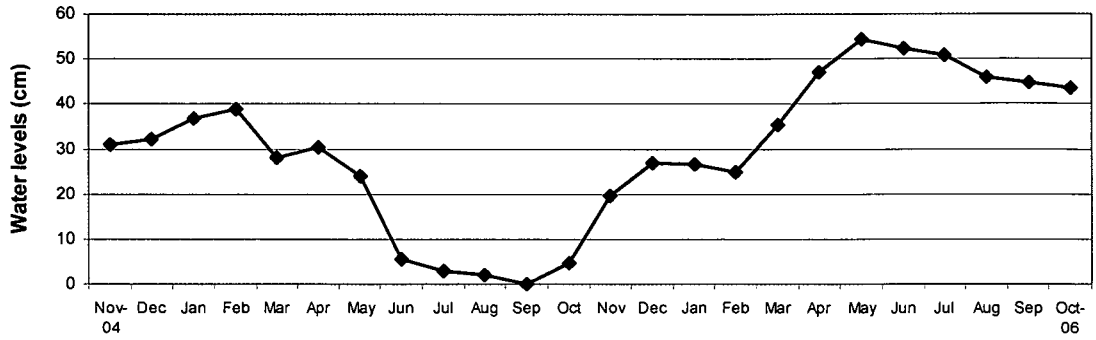


Figure 3.26 Changes of water levels in pool 6

Changes in water levels resulted in seasonal variation of water chemistry in the pool. Large changes of water chemistry can be clearly seen in winter and summer when water levels in the pool increased and decreased, respectively (*Figure 3.25*). Chemical variables especially nitrate and ammonia increased and showed peaks in winters when pool 6 received water from rain and runoff and at the same time when high numbers of wildfowl were present at the site (*Figure 3.27*). Similarly, TN also showed high values in winter and when water levels decreased and were low in other seasons. In summer, concentrations of ammonium nitrogen and phosphorus increased owing to release from the sediment.

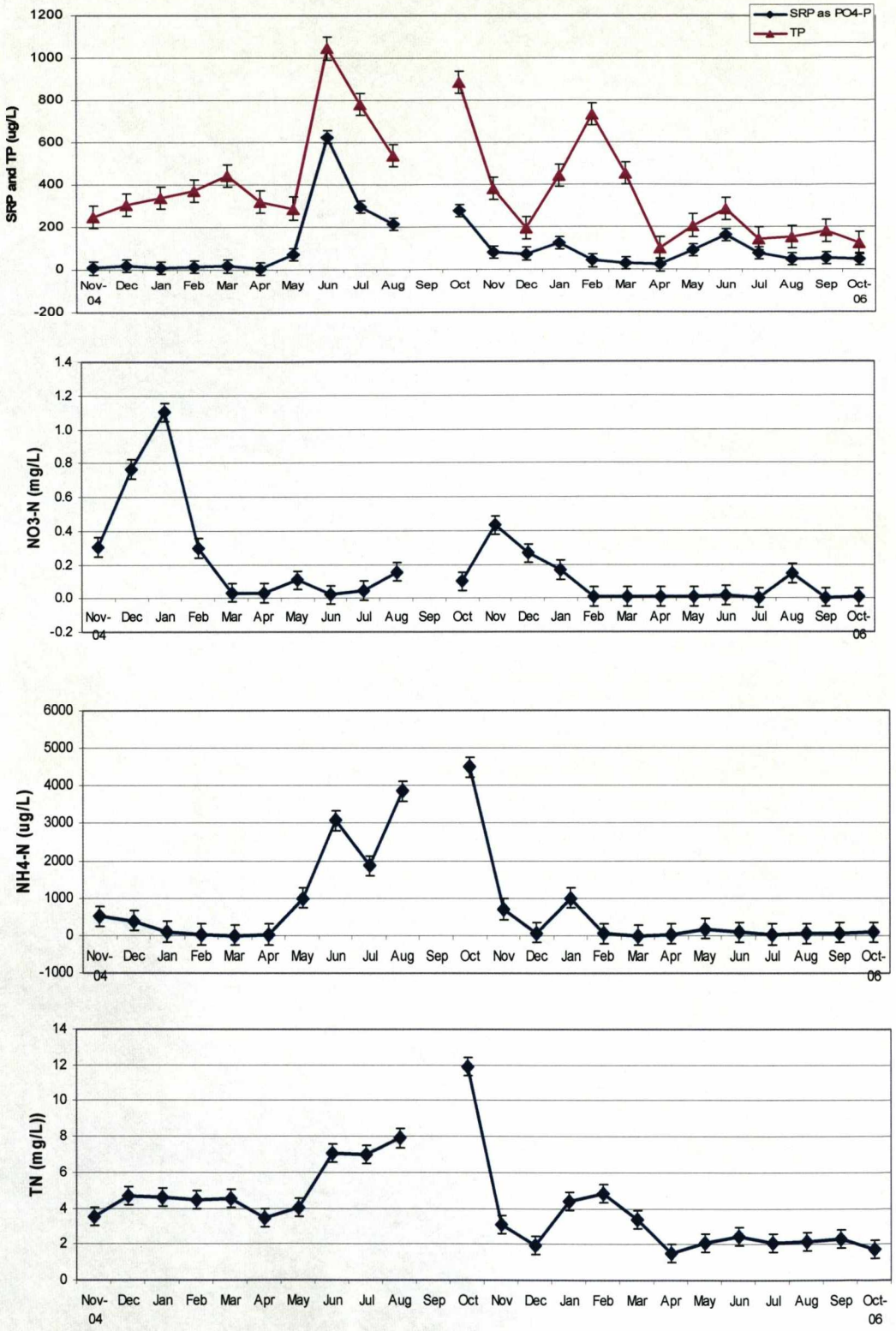


Figure 3.27 Seasonal changes of chemistry values in pool 6

Tukey analysis indicated seasonal significant differences of SRP (F = 3.380, $P = 0.04$) and NO_3^--N concentrations (F = 3.380, $P = 0.04$) (*Figure 28*). Specifically, NO_3^--N concentrations in winter were significantly different from those in spring ($P_{winter,spring} = 0.021$) and summer ($P_{winter,summer} = 0.036$). However, no significant differences in concentrations of TP (F = 0.54, $P = 0.661$), NH_4^+-N (F = 1.544, $P = 0.236$) and TN (F = 0.415, $P = 0.744$) between seasons were found.

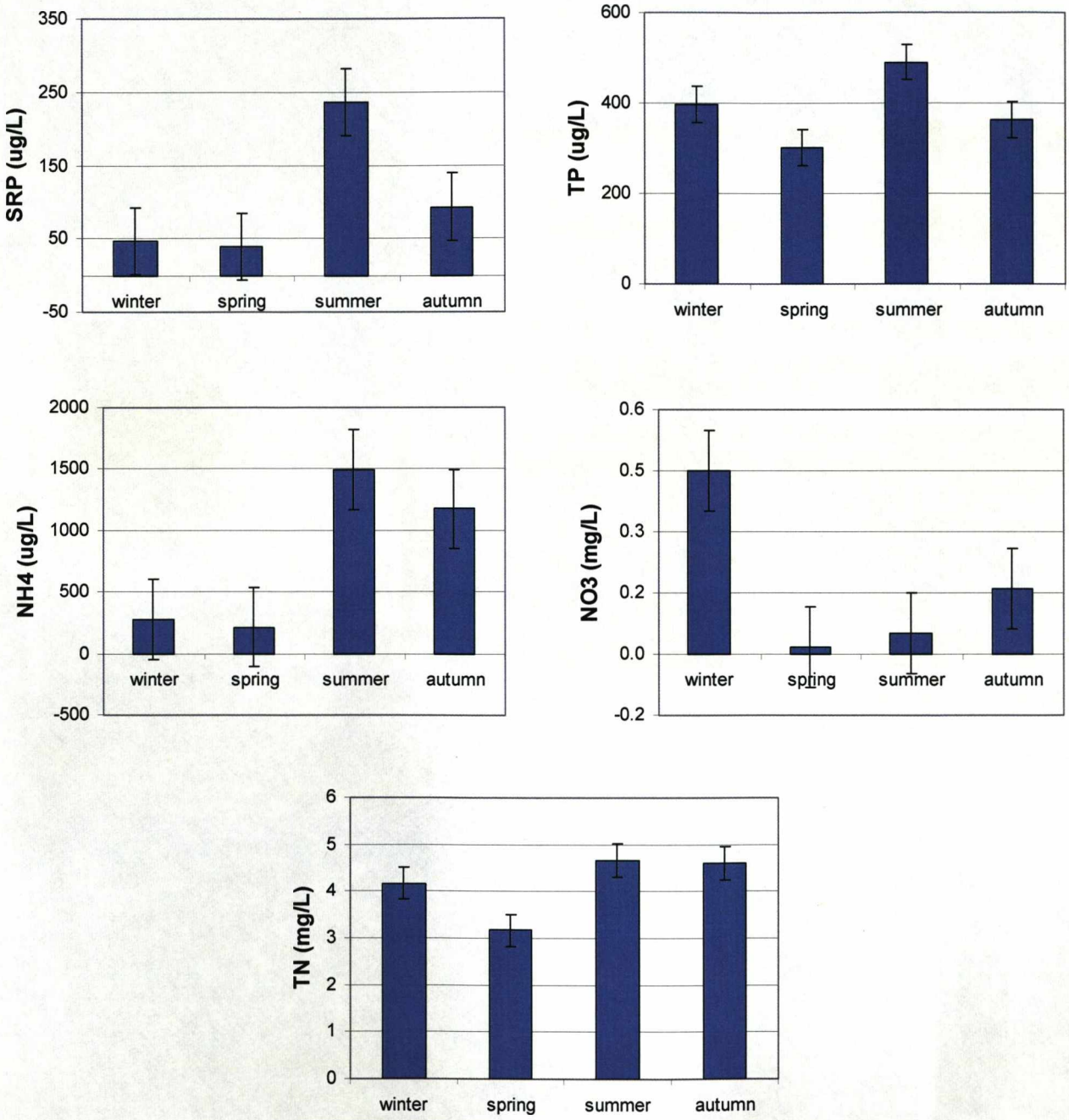


Figure 3.28 Average concentrations of nutrients in each season in pool 6 with standard error (n = 12)

Pool 6 was classified as a hypereutrophic pool based on Florida LAKEWATCH (2000), with high quantities of nutrients and chlorophyll a. TP and TN values were greater than 100 and 1,500 μg L^{-1}. Mean concentrations of SRP, TP, NH_4^+-N, NO_3^--N and TN (n = 49) in the pool were 104 ± 141 μg L^{-1}, 389 ± 254 μg L^{-1}, 773 ± 1,300 μg L^{-1}, 0.2 ±0.3 mg L^{-1} and 4.1 ± 2.4 mg L^{-1}, respectively. The TN/TP ratio was applied to determine whether phosphorus or nitrogen is the limiting nutrient in pool 6 and it was found that TN and TP ratio was 10.7, which was between 6.6 – 15.3 meaning that nitrogen or phosphorus could be limiting (Florida Lakewatch, 2000). TN and TP ratio in winter (10.5) and in the growing season (10.8) showed also similar results. Except in summer, the pool was nitrogen-limited as the TN/TP ratio was less than 10.

Our statistical analysis showed that overall both TP and SRP were significantly correlated with NH_4^+-N and TN, respectively. However, no significant correlations between concentrations of NO_3^--N and TP, SRP were found as shown in Table 3.10.

Table 3.10 Correlation coefficients of all nutrient variables from pool 6 at Brown Moss (n = 49)

Parameters	**TP**	**SRP**	**NO_3^--N**	**NH_4^+-N**
SRP	-			
NO_3^--N	-0.144	-0.247		
NH_4^+-N	0.724**	0.746**	-0.074	
TN	0.820**	0.564**	-	-

Remark; **. Correlation is significance at the 0.01 level and - auto correlated

I also tested correlations of nutrient concentrations in winter (December, January and February) and in the growth season (the rest) to see whether they were different (*Table 3.11*). The results revealed that no significant correlations among nutrients were found in winter. However, during the growing season, I found significant correlations between concentrations of SRP and TN and NH_4^+-N (*Table 3.12*).

Table 3.11 Correlation coefficients of nutrients in winter from pool 6 (n = 49)

Parameters	TP	SRP	NO_3^--N	NH_4^+-N
SRP	0.044			
NO_3^--N	-0.538	-0.538		
NH_4^+-N	0.017	0.748	-0.114	
TN	0.546	-0.473	0.352	0.060

Table 3.12 Correlation coefficients of nutrients in the growing season

Parameters	TP	SRP	NO_3^--N	NH_4^+-N
SRP	-			
NO_3^--N	0.004	-0.106		
NH_4^+-N	0.805**	0.730**	0.220	
TN	0.850**	0.624**	0.155	-

Remark; **. Correlation is significance at the 0.01 level and - auto correlated

3.3.6 Nutrient loading to pool 6

I estimated nutrient loading from all possible sources to pool 6 at Brown Moss and found that both external and internal loadings were strongly involved in nutrient addition to pool 6 and concentrations of nutrients stemming from each source varied with the seasons.

Birds, groundwater and rain were regarded as main external sources that supplied nutrients to pool 6. According to our estimation of external loading, I discovered that aquatic birds contributed the most phosphorus (73%) to pool 6 (*Table 3.13, Figure 3.29*). Groundwater and rain added 20% and 7% of phosphorus to the pool, respectively. In term of nitrogen, I found that groundwater is a major source that contributes up to 64% of nitrogen to the pool. Rain water and birds supplied 19% and 17% of nitrogen to pool 6, respectively.

Table 3.13. Amounts of external and internal nutrient loading to pool 6 at Brown Moss

Sources	**Phosphorus ($mg\ m^{-2} yr^{-1}$)**	**Nitrogen ($mg\ m^{-2} yr^{-1}$)**
External		
Birds	234 (73%)	741 (17%)
Soil/groundwater	63 (20%)	2,860 (64%)
Rain	24 (7%)	870 (19%)
% of total	**321 (100%)**	**4,471 (100%)**
External and Internal		
Birds	234 (12%)	741 (15%)
Soil/groundwater	63 (3%)	2,860 (59%)
Rain	24 (1%)	870 (18%)
Sediment release (internal source)	1,620 (84%)	340 (7%)
% of total (external and internal)	**1,941 (100%)**	**4,811 (100%)**

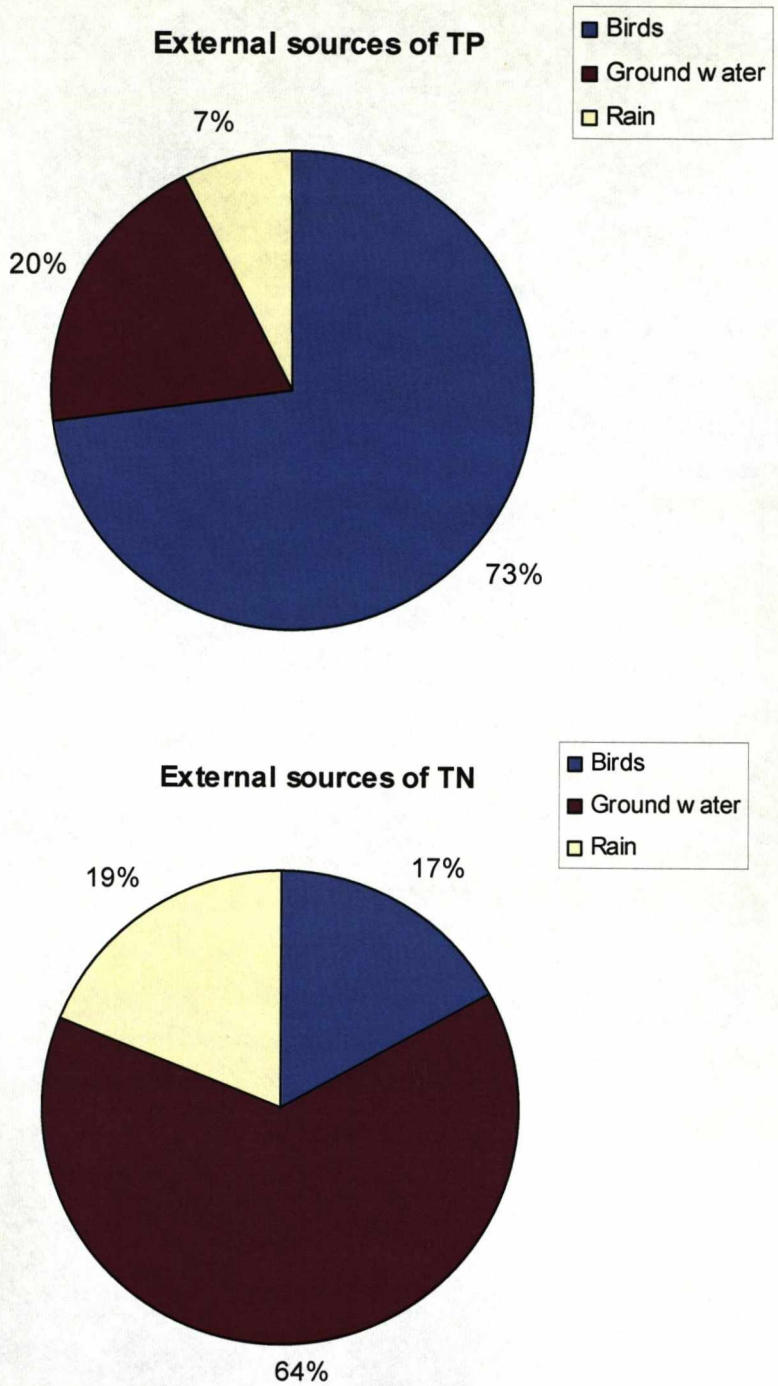


Figure 3.29 Contribution of TP and TN from external sources to pool 6 (P and N contribution by rain is less than 1%)

I also quantified nitrogen and phosphorus coming from the sediment which is the internal nutrient source of the pool and I compared percentage of nutrient contribution between internal and external sources. The results revealed that the sediment, an important internal source, is a major supplier of phosphorus and ground water contributed the most nitrogen to the pool. Phosphorus release from the sediment contributed up to 84% and ground water supplied 60% of nitrogen when compared with all external and internal sources combined. Birds, groundwater and rain added 12% and 3% and 1% of phosphorus and 18%, 15% and 7% of nitrogen came from rain, birds and sediments, respectively (*Figure 3.30*).

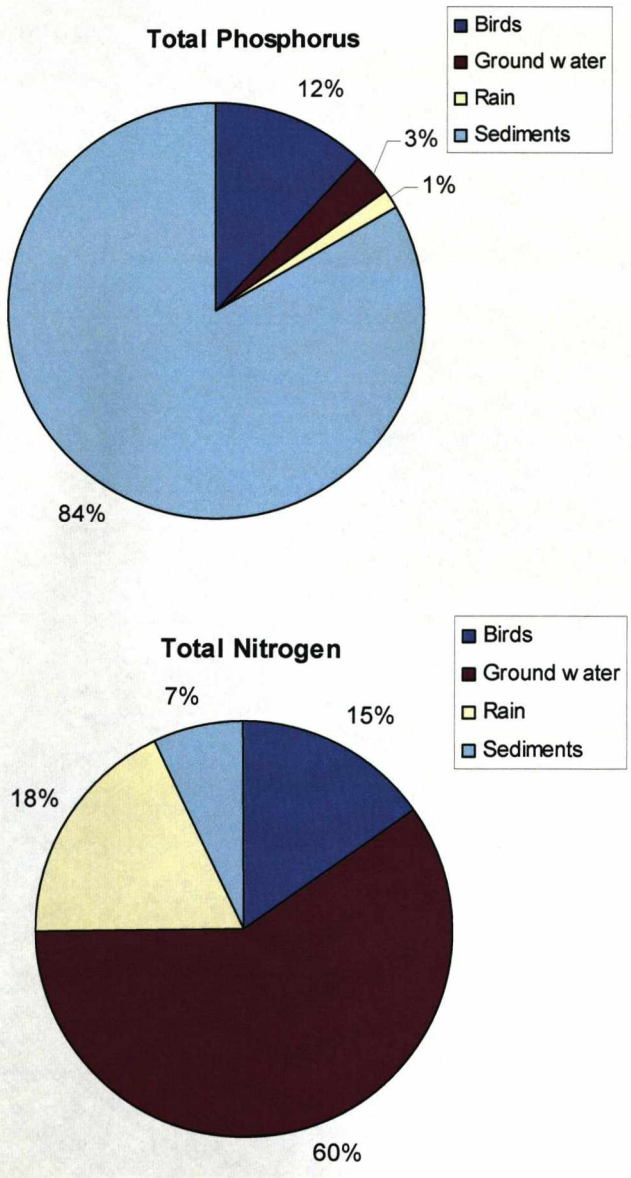


Figure 3.30 Percentage of nutrient loading from different sources to the pool

3.4. Discussion

3.4.1 Birds and water chemistry in pool 6

There was a variety of birds in pool 6 at Brown Moss (*Figure 3.31*). Water birds present at Brown Moss were most abundant and diverse in winter and less abundant in summer (*Figure 3.10*). Our results were similar to studies of Kitchell et al. (1999), Noordhuis et al. (2002) and Schmieder et al. (2006) in that the population of waterfowl in the pool was highest in winter. In summer when water levels started decreasing with lack of precipitation and high evaporation rate, number of waterfowl declined and there were only a few species of waterfowl, primarily mallard that still remained in the pool. Therefore, changes of season and water levels may affect numbers of waterfowl in pool 6, especially in 2004-2005. Savard et al. (1994) reported that the abundance of diving ducks was associated positively with pond depth and Suter (1994) stated that if pools are not too large and deep, waterfowl abundance should be a linear function of either surface area or shore length.

Between April and June, I found dabbling ducks and coot, which produced chicks, nesting near and in the pool, which was similar to Manny et al. (1994) who reported that young ducks and geese were found in Wintergreen lake in summer.

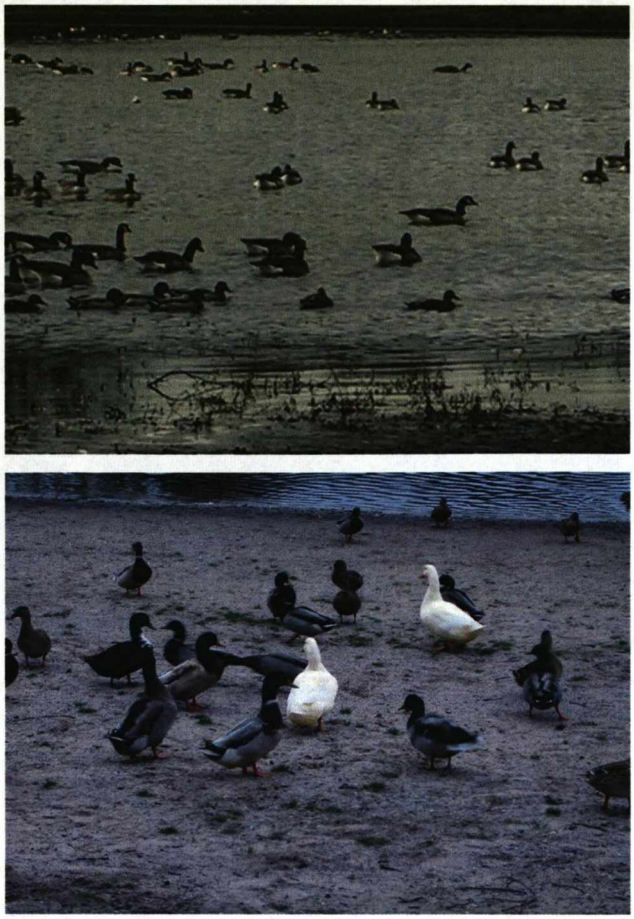


Figure 3.31 Congregation of Canada geese (above) and mallard and two domestic ducks (below) in the pool

Large water birds such as geese and swans disappeared when water levels fell. Change in numbers of Canada geese at Brown Moss is likely due to their movement among different water bodies in the area as they are non-migrating exotic species in the UK (Moss et al. 1996). Smaller sized ducks such as mallard and teal were, however, still found in pool 6, even though water level in summer was reduced.

Several studies have demonstrated that some species of waterfowl feed on land but rest on water (Raveling et al. 1972; Post et al. 1998; Woolhead, 1994). At Brown Moss, I found that water birds not only fed on land but also spent most of the time on or near the pool throughout the day. The pool could thus supply adequate diets such as aquatic plants and invertebrates to them (Afton and Hier, 1991; Afton et al. 1991). In winter during the day Canada geese were present everywhere in and around the pool, but at dusk they aggregated densely in large groups on the pool and it was assumed that this reflected a need for refuge from predators. Post et al. (1998) also explained

that birds dispersed themselves while at feeding time, consuming nutrients from larger area, then they congregated to roost in a more confined area.

According to our estimations of nutrient loading to the pool by birds, water birds were major external contributors of TP. They contributed about 73 % of TP and 17 % of TN to the pool (*Figure 3.27*). This was consistent with a recent study by Hahn et al. (2007) using a loading model in that water birds are only of minor importance for N loading into Dutch freshwaters but in contrast, P contribution is more important and may significantly affect the primary production of the often P-limited wetlands. Estimated loadings of TP and TN produced by waterfowl faeces were high in the autumn and winter months and low in spring and summer (*Figure 3.13*). Especially in autumn and winter, organic and inorganic loadings are believed to come from congregation of waterfowl such as Canada geese and mallard. Scherer et al. (1995) also found that P loading varied with the seasons (ie. with migration patterns) and the highest loading occurred in January, while the lowest loading rate occurred in May. Post et al. (1998) noted that peak loading rates occurred in mid-November.

No simple correlation between estimated nutrients excreted by waterfowl and nutrients measured in the water in pool 6 was found in the year-round study as birds are not the immediately dominant source of nutrients and they are abundant for only a few months in the pool in winter. There were other factors such as release of nutrients from the sediment in summer involved. However for a short period during winter in 2004 and 2005, I found strong correlation between bird abundance and water quality variables. I found that when waterfowl were present in high numbers in the winter months, ammonia in pool 6 increased. Thus, immediate increased ammonium nitrogen was related to numbers of birds. Donald et al. (1972) stated that significant quantities of nitrogenous compounds, particularly ammonia have been reported in the urine of domestic fowl and ducks and Stewart et al. (1969) found that the uric acid excreted by birds contributed 53.8%, the ammonia 29.2 % and the urea 1.5 % of the total nitrogen excreted. The three compounds together accounted for 84.5 % of the nitrogen excreted.

Several studies have shown that waterfowl faeces are an important source of nutrients to lakes as their droppings contain nitrogen and phosphorus (Manny et al. 1975; Moss and Leah, 1982; Sjaastad et al. 2003; Olson et al. 2005). Brandvold et al. (1976) who conducted a study in New Mexico, revealed that total nitrogen and phosphorus were found to increase downstream from where birds were roosting and this suggested the potential for the birds to contribute to eutrophication-related problems. Changes in Hickling Broad, a shallow, brackish lake in England from dominance by submerged aquatic plants to dominance by phytoplankton in the 1970s were also ascribed to the effects of guanotrophication by black-headed gulls (Bales et al. 1993; Irvine et al. 1993; Moss and Leah, 1982). Brinkhurst and Walsh (1967) suggested that droppings from black-headed gulls (*Larus ridibundus*) were primarily responsible for eutrophication in shallow Rostherne Mere. However, Carvalho (1993) showed that this was not the case. In conclusion, it is reasonable to assume that water birds are an important contributor of nutrients to many lakes (Manny et al. 1975; Harris et al. 1981; Manny et al. 1994; Kitchell et al. 1999; Olson et al. 2005)

It is also suggested that solid wastes excreted by water birds in winter may subsequently affect water chemistry in summer as it is seen by high concentrations of ammonium in summer that may be attributable to a lag phase in N release stored over the winter. Portnoy (1990) reported that over 56% of the annual P loading from gulls was deposited in a pond from mid-October to late December and this was consistent with Brandvold et al. (1976) and Moss et al. (1978) stating that the excreted uric acid of birds was insoluble and thus settled on the pond-floor, and through the decomposition process became the major nutrient source during the summer cyanophyte bloom. This was also consistent with Bales et al. (1993) and Hahn et al. (2007) who reported that besides a direct impact on lake productivity, avian contributed nutrients may enter the sediments and be released over longer periods. In summer, increased temperature helps accelerate decomposition of detritus and organic matter left over winter and therefore increased the amount of nutrients in the water (Stephen et al. 1997; Jensen and Andersen, 1992). In pool 6 at Brown Moss, phytoplankton increased rapidly and considerably at the same time when most birds were present in the pool. The growth of phytoplankton may be partly the result of droppings as they contain digested nutrients and the nutrients may be in a readily available form (Kear, 1963) that phytoplankton can take up.

3.4.2 Ground water and water chemistry in pool 6

Our finding was similar to several investigations in that ground water can be a significant source of nutrients to lakes (Loeb and Goldman, 1979; Brock et al. 1982; Shaw and Prepas, 1989a; Young and Briggs, 2005). Concentrations of nutrients in boreholes increased simultaneously when water levels in boreholes increased and this suggested that soil nutrients left were washed out through runoff and ground water after heavy rain. However, a lower water table in boreholes also resulted in high concentrations of nutrients perhaps owing to aerobic release from soil, high retention time and lack of washout.

There were differences between quantities of nutrients in boreholes near the fringes and in the middle area of the study site (*Figure 3.18*). Ground water near cropland and cottages was highly contaminated with ammonia, nitrate and phosphorus and this suggested that surrounding land use has a direct impact on nutrients in ground water (*Figure 32*). In borehole 1, installed near cottages, soluble reactive phosphorus was high and this may be the result of seepage from the cottages. In contrast, a lower concentration of phosphorus was detected in borehole 6 located near the pool. Although these fringing ground waters probably did not directly affect pool 6, they would influence pools to the margins of Brown Moss.

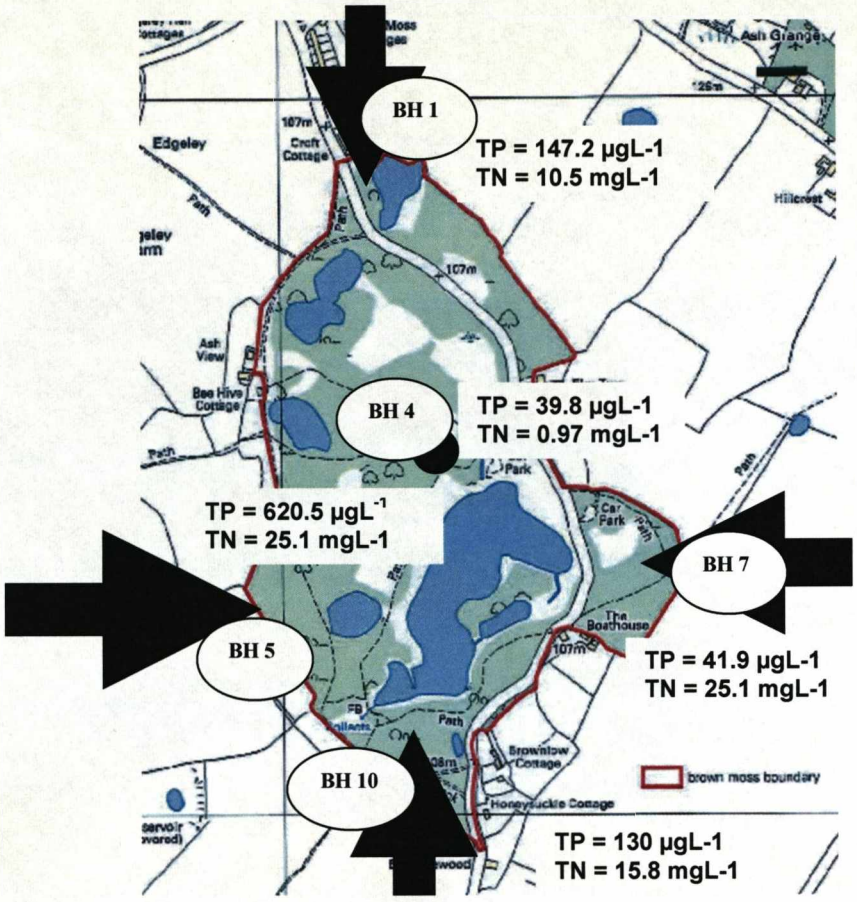


Figure 3.32 High concentrations of nutrients came from surrounding areas of Brown Moss (Selected boreholes were located at the fringe of the site)

I estimated nutrient loadings to the pool from groundwater and found that groundwater was a major source of nitrogen. Considered as an external source, it was estimated that groundwater supplied 64% of TN and 20% TP to the pool (*Figure 3.29*). Groundwater can be contaminated by domestic wastewater such as seepage from septic tank systems that potentially contribute to the ground water pollution (Canter and Nox, 1985). Porter et al. (1975) stated that septic tanks and associated disposal on fields can affect the quality of ground and surface waters as it is probable that many tanks leak into drains, ditches or directly into lakes and rivers. The report done by the Task Group Report (1996) also showed that leaching from septic tank systems significantly increased the concentrations of nutrients in wells from which homes obtained their water supply and Belanger and Mikutel (1985) revealed that groundwater seepage accounted for 14% of the total nutrient input to East Lake Tohopekaliga. Furthermore, Carvalho and Moss (1995) reported that a major cause of

the eutrophication in 35 lakes of 79 cases was sewage effluent with a further five possible cases of effluent problems. Boreholes 3, 5 and 10 all had the very high TP concentrations associated with external sources. Borehole 3 contributed to the nutrients of pool 6.

In addition, nitrogen and phosphorus can leach from agricultural land to ground water and pose a threat to water quality (Jiao et al. 2004). High concentrations of total nitrogen and nitrate were detected in boreholes 5, 7 and 10, which are near intensive farmland (*Figure 3.33*) and this suggested that manures and fertilizers used in agricultural areas may be carried by surface run off and then to groundwater.

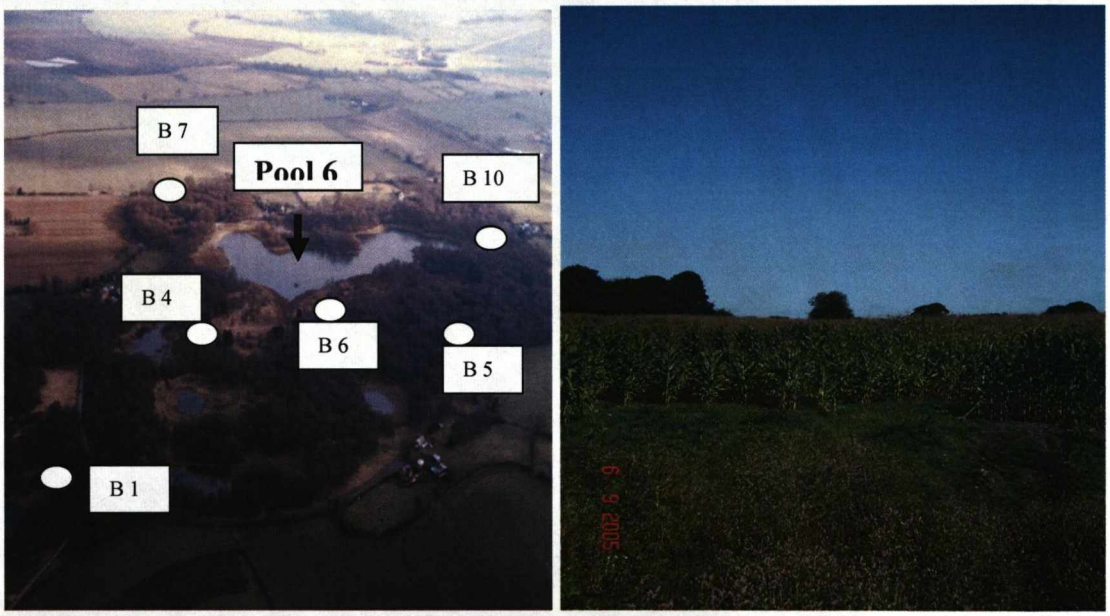


Figure 3.33 Types of land use around Brown Moss and locations of boreholes (left) and intensive cropland on the east of the site (right)

As mentioned earlier, groundwater is a major contributor of nitrogen to pool 6 at Brown Moss. In winter 2005 (November-December), nitrate in pool 6 increased at the same time as that in boreholes and this suggested that pool 6 is probably affected by nutrients from ground water. Especially on the east side, nitrate can potentially enter the water body during the pouring rain because the location of cropland is not far from the pool and is on a slope, compared with that on the west side. Moss et al. (1997) and Porter et al. (1975) stated that in agricultural catchments, nitrate added as chemical and biological fertilizers is very soluble and is easily washed out to the streams. In

England and Wales, approximately 58,000 tonnes of nitrate nitrogen are lost to waters each year (Grimble, 1994). Verchot et al. (1997) also maintained that subsurface flow often represents the major pathway for movement of dissolved nutrients such as nitrate from cropland and groundwater nitrate contamination typically appears in localised areas and is often related to intensive agricultural or urban uses on coarse-textured soils (Poe et al. 1998). Therefore, the effect of groundwater on pool 6 can be significant and groundwater and associated nutrients can enrich the pool and stimulate the growth of phytoplankton. Borehole 8 appears to contribute a significant amount of nitrogen to the total entering the pool.

However, buffer zones may reduce concentrations of nutrients in subsurface water and run off. Our evidence demonstrated that lower concentrations of nutrients were detected in boreholes in the middle of the site surrounded by woodland (*Figure 29*). Lowrance et al. (1989) and Barling and Moore (1994) explained that riparian forests are nutrient filters. Buffers are often used in agricultural watersheds for management. As nutrients, especially nitrate, are washed out and carried downslope from agriculture areas toward pool 6, they are taken up by trees and shrubs in woodland and some are lost by denitrification during transportation (Brinson et al. 1984; Haycock and Pinay, 1993; Young and Briggs, 2005). Nutrients such as phosphorus can also be absorbed by physical reactions onto clay surfaces by complexing with calcium, aluminium and iron (Logan and McLean, 1973; Loeb and Goldman, 1979). Thus, biological purification and filtration by buffer areas can effectively reduce concentrations of nutrients in contaminated water before entering the pool.

Furthermore, a study of the Little River watershed in the Tifton upland, a sub province of the Georgia coastal plain, revealed that despite large fertilizer inputs to row crop fields, stream flow outputs of nitrate from the watershed were less than input in precipitation. It was hypothesised that the riparian ecosystem filtered nutrients and helped maintain water quality on this agricultural watershed (Amussen et al. 1979). Doyle et al. (1977) evaluated the effectiveness of forest and grass buffer strips in improving the water quality of manure-polluted runoff and found that forests and grass buffer strips produced significant reductions in nutrient levels, particularly in the first few meters. They discovered reductions of 94.7% and 99.7 % of nitrogen and phosphorus, respectively after 3.8 m in the forest buffer strip.

3.4.3 Rain water and water chemistry in pool 6

Higher concentrations of nutrients in rain water were detected in summers and low concentrations were recorded in winters. Our results were consistent with many studies in that the seasonal pattern of atmospheric P and N concentrations is inversely related to the precipitation (Shaw et al. 1989; Bergametti et al. 1992; Migon and Sandroni, 1999; Rashid et al. 2005). Shaw et al. 1989 reported that rates of atmospheric loading of TP and TN are highest from May to August and are very low during winter. High atmospheric deposition during May may be the most important external source of nutrients to lakes in central Alberta. Low amounts of rain fall and high anthropogenic emissions from industrial and agricultural activities may result in higher mean concentrations of nutrients in summer. In contrast, lower mean concentrations of nutrients in rain water in winter may be likely due to extreme dilution of nutrient-based gases from the atmosphere (Bootsma et al. 1999).

Berganetti et al. (1992) explained that during summer, rain events are scarcer and the residence time of atmospheric particles is increased, and in this case the few rain events occurring are more efficient in washing out a highly loaded atmosphere. On the contrary, during winter, rain events occur more often and this leads to a scavenging of aerosol particles emitted, resulting in little atmospheric particulate matter to be scavenged later.

Major sources of the nutrients in rain water are anthropogenic, especially for nitrate and ammonium (Herut et al. 1999). Ferm (1998) and Prospero et al. (1996) reported that the present-day emissions of NH_x from anthropogenic sources are about four-to-five times greater than natural sources, mostly due to the impact of emissions from animal wastes associated with food production while non-atmospheric sources supply most of the phosphate (Herut et al. 1999). I did not have data on concentrations of nutrients in rain water at Brown Moss but I expected that nutrients, especially ammonia in rain water at Brown Moss may be relatively higher than that measured at the University of Liverpool. This is because fertilizer use, manures and wastes excreted by animals in agricultural areas and grazed pastures surrounding the site are potential sources of emissions. The report by the European Centre for Ecotoxicology and Toxicology of Chemicals (ECETOC, 1994) clearly indicate that agriculture is the

main source for ammonia emissions and on average in Western Europe, 92% of all ammonia originates from agriculture. Manure was used in agricultural areas around Brown Moss as I sometimes saw and smelled fresh manures while sampling around the site. On the other hand nitrate, contributed from automobile exhaust gases oxidising in the atmosphere, was high to have been relatively greater in Liverpool.

Nutrients and pollutants in the atmosphere can be delivered to ecosystems by atmospheric deposition. Lovett (1994) explained that wet deposition is the deposition of material contained in precipitation and it is an important form of atmospheric deposition for ecosystems. Wet deposition results from the incorporation of atmospheric particles and gases into cloud droplets and their subsequent precipitation as rain or snow, or from the scavenging of particles and gases by rain drops or snowflakes as they fall. Accordingly, excessive atmospheric pollutants contributing to nitrogen deposition can be deposited and cause eutrophication in freshwater and marine ecosystems (Fowler et al. 1989; Lovett and Kinsman, 1990; Groot Koerkamp et al. 1998; Kirchmann et al. 1998).

Several investigations have revealed that atmospheric deposition contributes significant amounts of nutrients to lakes and oceans (Caiazza et al. 1978; Shaw et al. 1989; Jassby et al. 1994; Kirchmann et al. 1998; Rashid et al. 2005). For example, Rashid et al. (2005) stated in their study that high nutrient deposition rates were found on the African Great lakes as a consequence of human activities. Shaw et al. 1989 also found that average rates of atmospheric deposition of TP and TN to Narrow Lake in central Alberta were 20 and 424 mg m^{-2} yr^{-1} which were similar to TP and TN loading to Brown Moss estimated as 24 and 870 mg m^{-2} yr^{-1}, respectively.

I found a weak relationship between amounts of nutrients in rain water and in pool 6 and this is similar to Wambach and Mallin (unpublished data) revealing that precipitation was correlated with monthly water quality parameters in Greenfield Lake, North Carolina. I estimated external contribution of nutrients from rain to pool 6 and discovered that rain water contributed 7% and 19% of TP and TN to the pool compared with other external sources (*Figure 3.27*). Relatively high percentage of nutrients, especially nitrogen in rain water suggested that atmospheric wet deposition is an important nutrient source of pool 6. However, rainfall may cause its greatest

impact on nutrient addition through surface run off. Increases in nutrients in pool 6, related to rainfall, are probably mostly due to increased loading rates from soil and surface inflows (Shaw et al. 1989).

3.4.4 Release of nutrients and water chemistry in pool 6

Our experiments showed that nutrients were both released and taken up by sediments as also indicated in the studies of Cerco (1989) and Selig and Schlungbaum (2003). Nutrients, especially ammonium nitrogen and phosphorus mostly increased in summer and were low and taken up in other seasons. This result was consistent with Ekholm et al. (1997) and Sondergaard (1999) stating that phosphorus concentrations increased during summer and were higher than in winter in lakes. Our finding was also similar to the study of Sondergaard (1999) who stated that TP in most Danish lakes increases during summer and summer TP concentrations depend on internal loading. The release of nitrate into the pool appears to be negligible in comparison with SRP and ammonia.

Nutrient release rates varied throughout the season. In summer, nutrient release rates were high, and low in winter. Maximum phosphorus and ammonium nitrogen release rates recorded were 19 mg P m^{-2} d^{-1} and 53 mg N m^{-2} d^{-1} in summer. Similar results for nutrient release rates are found from other eutrophic/hypereutrophic lakes. For examples, phosphorus release from undisturbed sediment sampled in May in eutrophic Lake Arreso, Denmark was 12 mg P m^{-2} d^{-1} (Sondergaard et al. 1992). The results of Phillips el at. (1994) at Cockshoot Broad showed that up to 26 mg P m^{-2} d^{-1} of phosphorus was released and by Ensenat (2000) revealed that maximum ammonium nitrogen released at Lake Okeechobee was 40.1 mg N m^{-2} d^{-1}. Graetz et al. (1973) studied nitrogen dynamics in four eutrophic Wisconsin lakes and reported that under anoxic conditions, sediments released ammonia at rates ranging from 12 – 50 mg N m^{-2} d^{-1}. In Spremberg Reservoir, Germany, the mean phosphorus and nitrogen release rates were 9.7 mg P m^{-2} d^{-1} and 43.4 mg N m^{-2} d^{-1}, respectively (Kleeberg and Heidenreich, 2004).

The release of nutrients, especially phosphorus and ammonium nitrogen over 24 hours was consistent with high quantities of nutrients measured in pool 6 in summer. Thus, it is likely that the summer phosphorus and nitrogen concentrations in pool 6 are controlled by the internal rather than external load (Phillips and Jackson, 1990 and Pettersson, 1998), which was consistent with Scherer et al. (1995) stating that loading rate occurring in summer appeared to be from nutrients derived from the lake itself, thus representing nutrient cycling. Estimation of internal nutrient loading from the sediment also showed that the sediments are the main source of phosphorus. According to our estimation, the sediment contributed up to 84 % of total phosphorus and 7% of total nitrogen, compared with external sources. Knuutila et al. (1994) stated that the summer internal phosphorus load was estimated to be as high as two to threefold the external load, which is consistent with Reddy et al. (1996) who found that internal nutrient loads may be equal to or higher than external loads.

Increase of nutrients during summer is likely due to mineralisation of organic matter. Biological activities of micro organisms are one of main explanations for regulating nutrient cycling in lakes and living micro organisms can play a direct role by acting as either a sink or a source for phosphate (Bostrum et al. 1982; Petterson, 1998). Mineralisation is an important process, by which organic compounds are transformed, usually by bacteria, to inorganic substances such as phosphate and ammonium (Porter, 1975). Especially in aquatic systems, bacteria are the major decomposers of organic matter (Fenchel and Blackburn, 1979). Due to mineralisation, therefore prerequisites for a high phosphorus release from sediment are the most likely to be found in the sediment (Bostrom and Pettersson, 1982).

Because mineralisation processes mainly rely on sediment bacteria (Pettersson, 1998), temperature is one of key factors involved in regulating nutrient release (Jensen and Andersen, 1992). During winter, mineralization is slow because of cold conditions. But in summer when temperature is higher, decomposition of detritus is accelerated (Stephen et al. 1997; Sondergaard, 1999) and as a result nutrients are released into the water column (Jensen and Andersen, 1992). Equations representing transformation of organic compounds to inorganic substances (Porter, 1975) are presented in the next page.

$$\text{Organic nitrogen} \xrightarrow[\text{Ammonification}]{} NH_3 \xrightarrow[\text{Hydrolysis}]{} NH_4^+ + OH^-$$

$$\text{Organic phosphorus} \xrightarrow[\text{Mineralisation}]{} H_2PO_4^- / H_2PO_4^{2-}$$

Microorganisms such as bacteria not only help release nutrients but also take up nutrients. Laboratory experiments of Gachter et al. (1988) showed that sediment micro organisms can rapidly take up soluble reactive phosphorus in the form of polyphosphate. However, rates of phosphorus release overall in pool 6 were obviously higher than such uptake by micro organisms, especially in summer.

Additionally, nutrients released from the sediment in pool 6 Brown Moss may also come from chemical mechanisms. Chemical processes play a major role in regulating nutrient cycling in the sediment in lakes. Mortimer (1941,1942), Holden and Armstrong, (1986), Gachter, (1988), De Groot, (1991), Moss et al. (1996) and Sondergaard et al. (2003) noted the important chemical transformation of nutrients related to redox conditions between the surface sediment and overlying water in fertile conditions in lakes. When sediment becomes anoxic and sediment redox potential decreases, ferric complexes dissolve, [Fe(III)) is reduced to Fe(II)] and subsequently phosphate previously sorbed with iron (III) is released into solution and remains in high concentrations (Mortimer, 1941,1942). In anoxia, sulphate is also transformed to sulphide and it combines with iron (II) in the sediment to form the insoluble iron sulphide (FeS), giving intense blackness in the sediment. A chemical reaction in water and sediments that may control solubility of phosphorus and form insoluble iron sulphide is presented below (Murphy et al. 2001). Blackness of surface sediments in the study pool suggested anaerobic conditions and thus it was believed that nutrients released partly came from this chemical mechanism.

$$3S^- + Fe_3(PO_4)_2 \longrightarrow 3FeS + 2PO_4^{-3}$$

Nitrate concentration also regulates release of phosphate. Andersen (1982) and Jensen and Andersen (1992) explained that oxidised nitrogen buffers the redox potential of the surface sediment and prevents a release of phosphate. This is consistent with our experiments in that in winter, nitrate concentration stemming from soil and shallow groundwater was high and this may reduce P release because nitrate, by oxidising the sediment, increases the thickness of the aerobic layer at the sediment surface that inhibits the release of nutrients. Jensen and Andersen (1992) also revealed that during winter high nitrate concentrations increased the sorption of phosphate by the sediments.

Under anoxic conditions, ammonia is also released from the sediment. Rysgaard et al. (1994) explained a mechanism that is responsible for the release of ammonia from anoxic sediment. In anoxic sediment nitrification, the bacteria may convert ammonia to nitrite but often the process is not completed and as a result nitrate formation is inhibited because of lack of oxygen. Therefore ammonia converted from organic nitrogen substances is released into the overlying water. Steingruber et al. (2001) also showed that high rates of ammonia regeneration come from decomposition of organic matter. When oxygen is present, nitrification of ammonia can proceed (Beutel, 2006). But because of the absence of oxygen in sediments, nitrification, the oxidation of ammonia into nitrite, and the subsequent oxidation to nitrate performed by bacteria, does not take place and as a result, nitrate is not produced and that is one reason why release of nitrate from the sediment was not detected. Nitrate can also be reduced to gaseous nitrogen and lost from aquatic ecosystem via denitrification in anoxic conditions in the sediment (Rysgaard et al. 1994; Steingruber et al. 2001).

The last process involved in nutrient cycling in the sediment concerns physical mechanisms and this process may also help nutrients to be released from the sediment in the pool at Brown Moss. Fan et al. (2001) stated that resuspension by wind or animals can also lead to increased release rates of phosphorus especially in areas shallower than 3 m (Huttula, 1994). This was accordance with the study of Sondergaard et al. (1992) stating that the P release caused by sediment resuspension was 20-30 times greater than the release from undisturbed sediment cores. They explained that SRP-release during resuspension is mainly determined by the SRP equilibrium conditions and proceeds until equilibrium concentrations are reached.

I found that the population of benthic macroinvertebrates such as *Chironomus* increased in summer (see details in Chapter 4) and this may cause bioturbation of the sediment to increase rates of nutrient release too (Graneli, 1979; Petr, 1977; Fukuhara and Sakamoto, 1987; Andersson et al. 1988). Bostrom et al. (1982) and Phillips et al. (1994) showed that activities of *Chironomus* increase the release of phosphorus due to an increased water exchange between anaerobic water layers and lake water.

3.5 Conclusion

Our results demonstrated that internal and external nutrient loads seasonally corresponded well with changes in water chemistry in pool 6 although no simple and long term correlation between water chemistry in the pool and loadings from each source was found. Furthermore, nutrient loading from different sources affected water chemistry in pool 6 on different occasions.

Nitrogen and phosphorus concentrations in pool 6 were maintained in winter by external loading mostly from birds, shallow ground and soil water and rain. Gathering of large numbers of waterfowl in winter did immediately increase nitrogen compounds and phosphorus in the pool and this may activate growth of phytoplankton. Moreover, droppings from birds appeared to enrich sediments in winter and in summer these organic contents can become a major source of internal loadings. Phosphorus input into pool 6 in winter mainly comes from birds.

Groundwater is also considered an important source of nutrients, especially nitrogen to pool 6. Nitrate nitrogen in shallow groundwater in boreholes increased mostly in winter and was low in other seasons and this suggested that high quantities of nitrogen in ground water in winter could be transported by rainfall and sub-surface flow. Manure and fertilisers from agricultural areas are considered the main sources of nitrogen in ground water. Phosphorus concentrations were high in boreholes near cottages and this indicated the impact of seepage of domestic sewage in groundwater. However, pool 6 was less affected by land use and groundwater because the buffer areas on the south of the pool helped purify polluted shallow groundwater. Without

buffer areas, pools at Brown Moss could have been more and severely affected by contaminated surface runoff and ground water.

The results also clearly showed that direct rainwater is a major source of nutrients, especially nitrogen. Rain plays two major roles in nutrient enrichment. The first is that nutrients are directly added into the pool from rainfall and the second role is that rain washes out nutrients on land into groundwater and then nutrients are transported to the pool eventually.

During summer, external load was relatively small and was less important in nutrient input into pool 6. This is because there were fewer birds in summer and lower amounts of net rainfall and surface run off adding nutrients to the pool. In contrast, phosphorus internal load became crucial through sediment release. During winter, the sediment acts as a sink due to high assimilation rates from external sources and unfavourable weather conditions but in summer the role of sediment is changed and sediments become an important source as indicated by high amounts of nutrients released from the sediment. Sediments released both phosphorus and nitrogen in summer due to extensive mineralisation and physical and chemical mechanisms. Summary of seasonal nutrient input to pool 6 is shown below.

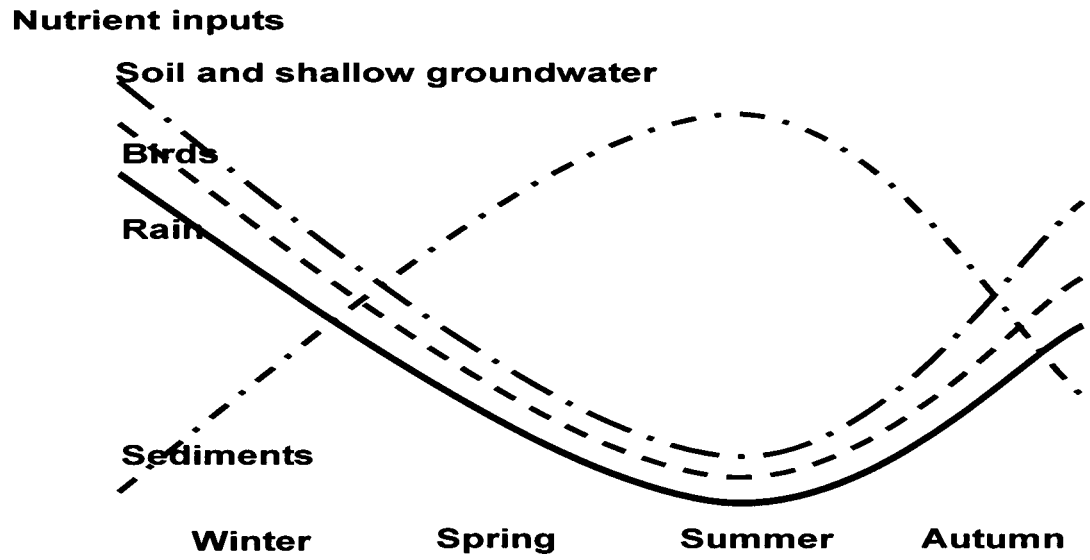


Figure 3.34 Seasonal nutrient input to pool 6 at Brown Moss

Chapter 4 Sediment composition and nutrient release in a shallow lake

Chapter 4

Sediment composition and nutrient release in a shallow lake

4.1 Introduction

The sediment in lakes plays a crucial role in ecosystem function. Sediments comprise organic and inorganic materials that come from external and internal sources (Bostrom et al. 1988; Sondergaard et al. 1993; Lynette et al. 2004; Nowlin et al. 2005). External material of sediments in lakes comes from soil erosion and particles of minerals and rock fragments and allochthonous leaf litter that are washed out into lakes from the catchment. The internal sediment is originally generated from external material and from excretion of aquatic animals and decaying flora and fauna (Andersson et al. 1988; Graneli and Solander, 1988; Petterson, 1998).

The sediment is important for aquatic plants and animals. It provides the substratum for rooted aquatic plants and periphyton and different sediment conditions influence the species composition and variety of submerged aquatic plant communities (Barko and Smart, 1984). In the littoral zone, sediments are usually firm, structured and suitable to support aquatic plants; in turn, marginal plants help protect sediment and banks from erosion and collapse. In deeper areas, sediment is loose, amorphous and uniformly fine (Hearn et al. 2002). Nutrients available in sediments are also important for aquatic macrophytes and may help keep lakes in a clear state dominated by plants by providing a phosphorus source more or less exclusive to them. Experiments with isotopes of nitrogen and phosphorus have shown ample uptake from the sediment by even small root systems and translocation to the shoots of aquatic plants (Denny, 1980).

The sediment also supports a wide range of benthic communities such as bacteria, protozoa, small crustaceans associated with plant debris, oligochaete worms and other animals (Moss, 1998). A crucial source of energy for benthic secondary production stems from lake sediments. Decaying organic material and detritus stored in the mud are food sources for micro organisms and macro invertebrates such as benthic worms and insect larvae. Benthic worms especially are scavengers and play an important role by contributing to the decomposition of dead animal and plant remains. Sedimentary

bacteria are also involved in biodegradation by breaking down organic material and regenerating nutrients (Capone and Kiene, 1988; Fenchel and Blackburn, 1979). As a result of decomposition, sediments are sources of nutrients for aquatic plants and microscopic algae (Ristora et al. 1999; Swan and Palmer, 2000).

In addition, sediments form continuous records of past events in the lake and catchment and, thus, they have historical and archaeological values. When organisms in lakes moult or die, they leave remains. Remains of aquatic flora and fauna such as pollen grains, spores, diatom and cladocerans tend to stay in the sediments on site and these can be used to reconstruct the past history of the lake and its surroundings (Frey, 1988; Hearn, 2002). Schelske and Hodell (1991, 1995) proposed that the carbon isotopes of bulk sedimentary organic matter in lakes could be used to reconstruct the history of nutrient loading and eutrophication and to trace the response of a lake to nutrient loading. Sediment data may also be used to confirm evidence for past catchment disturbances, or to quantify the soil losses under natural and accelerated forms of erosion (Dearing, 1991; Foster et al. 1990).

A variety of physical, chemical and biochemical processes also takes place in lake sediments, for example those associated with sorption processes, mineral-water equilibria and biodegradation (Hemond and Fechner-Levy, 1999; Kramer, 1964; Montigny and Prairie, 1993; Mortimer, 1971). These mechanisms in lake sediments are important and reflect conditions in the water column (Binford and Brenner, 1986).

Moreover, several studies have indicated that the bottom sediments may either act as a sink or a source of N and P nutrients in lakes since nutrients can be both released from or bound in the sediment (Jensen et al. 1992; Phillips et al. 1994; Sondergaard et al. 2001; Beutel, 2006). Sediments are considered a source of nutrients when N and P concentrations are released through decomposition of organic material and through physical and chemical mechanisms (Gachter et al. 1988; Sinke et al. 1990; Jensen and Andersen, 1992). Release of nutrients from the sediment to the epilimnion can stimulate phytoplankton growth and degrade surface water quality in a process known as internal loading (Pettersson, 1998; Welch and Jacoby, 2001; Beutel, 2006). In contrast, the sediments may become a sink if nutrients are bound in the sediment via biogeochemical processes (Wulff et al. 1990; Reddy et al. 1996). Serving as a sink,

the sediment may control the trophic state of the aquatic system and limit productivity of lakes (Conley et al. 1997). There are many factors that influence cycling and status of nutrients in lake sediments.

This chapter is concerned with roles of sediments in pool 6 at Brown Moss. I investigated composition and chemistry of the sediment from the surface to a depth of 15 cm to gain an insight into the importance of pool sediment on present nutrient status. I also quantified key elements in the sediment that might lead to the understanding of nutrient regulation and cycling in the pool. Moreover, I measured release of nutrients from the areas where the sediment composition was studied to investigate correlation between nutrient release and chemical variables in sediments and to determine whether sediments are functioning as a sink or source for N and P in pool 6.

4.2 Method

4.2.1 Sediment composition

To study chemical composition of pool sediments, I took three replicate sediment samples from five locations (A, B, C, D and E) around pool 6 at Brown Moss in November 2006, using plastic core samplers (length 50 cm diameter 6.9 cm) (*Figure 4.1*). These were the same cores as collected for Chapter 3.

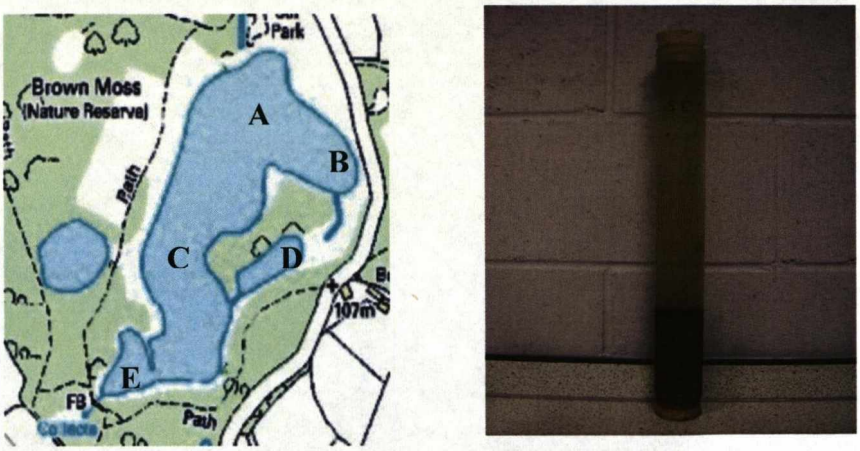


Figure 4.1 Sampling stations at Brown Moss (left) and a plastic core sampler used to collect the sediment from the pool (right)

After collection, samples were taken back to the laboratory at the University of Liverpool. The water was siphoned off and then a plunger was used to push the sediment slowly from the bottom of the cores until the surface sediment reached the top. The sediments were then sliced into three sections (0-1 cm, 1-5 cm, 5-15 cm) and altogether 45 sediment samples were analysed. Approximately 10g of material from each stratum was weighed into a pre weighed crucible for subsequent analysis.

To measure water content of the sediment, samples were dried at 105 ^{o}C for 24 hours and weighed (Sondergaard at al. 1993). Loss on ignition or LI (measurement of organic content of sediments (Boyle, 2001; Beaudoin, 2004) was measured by burning samples at 400 ^{o}C for three hours and reweighing the crucibles. I subsequently burned samples at 900 ^{o}C for four hours for measurement of carbonate content and I again reweighed the crucibles. Calculation of loss of moisture content, ignition and carbonate content was based on Department of Geography, (2005), Allen, (1989) and Trautmann and Richard, (2006).

$$\text{Calculation of moisture content} = \frac{W_W - Ws \times 100\%}{W_W}$$

$$\text{Calculation of loss on ignition} = \frac{W_S - W_A \times 100\%}{W_S - W_C}$$

$$\text{Calculation of Carbonate} = \frac{[(W_A - W_{CA}) \times 1000] \times 1.36 \quad (\text{mg g}^{-1})}{W_S - W_C}$$

In which:

W_A = weight of the sample after burning at 400 °C

W_C = weight of crucible

W_S = weight of the sample after drying at 105 °C

W_W = wet weight of the sample

W_{CA} = weigh of the sample after burning at 900 °C

1.36 is obtained from molecular weight of CO_3/CO_2

For analysis of iron (Fe), total phosphorus (TP) and calcium (Ca) contents, I used dried ash samples previously burned at 900 °C for four hours (Andersen, 1976). Dried ash samples (0.25 g) were dissolved in 10 ml of 2 M hydrochloric acid in glass tubes and left overnight (12 hours) for extraction. The following day, each sample was made up to 25 ml with deionised water and shaken well in a rotamixer before being filtered through Whatman GF/C (pore size 1.2 μm) filters. Extraction solutions were then analysed spectrophotometrically (Jenway 6405 UV/VB) for total phosphorus following the molydate reaction (Mackereth et al. 1989) while Fe and Ca were measured by atomic absorbance (UNICAM 929). Detection limits of ash samples for phosphate, Ca and Fe were 0.1, 1 and 0.2 mg g^{-1}, respectively.

4.2.2 Nutrient release

Measurement began in November 2004 and ended in October 2006. Undisturbed sediments and water (with triple replication) were collected monthly by plastic core samplers (length 50 cm diameter 6.9 cm) from the same five locations in pool 6 mentioned earlier to examine release of nutrients from the sediment (*Figure 4.1*). The core samples were sealed bottom and top with rubber bungs and kept in dark in an opaque tube and brought back to the laboratory at the University of Liverpool (*Figure 4.2*).

Figure 4.2 Plastic core samplers were kept dark in opaque tubes.

Methods for determining nutrient release rates are described in Chapter 3, section 3.2.4.

Tukey honestly significant difference (HSD) test was performed to distinguish differences of chemical values between surface, middle and bottom sediment and rates of nutrient release among sites. I also used Pearson correlation coefficients (r) to investigate correlation of chemical properties in the sediment and between chemical variables in the sediments and nutrient release. SPSS for Window 15 performed all statistical analysis.

4.3 Results

4.3.1 Sediment composition

Water content was highest in top layers of all samples and the sediment was drier at greater depths (*Figure 4.3*). Most fresh sediment samples were black but the colour of sediments taken from site B was slightly brown. Particles in all samples were fine and I found some pieces of organic matter such as roots and leaves in some samples. At site B, the lower layers of the sediment were mainly sand and gravel and the sediment layer was thin compared with that at other sites around the pool where particles had accumulated to greater thickness.

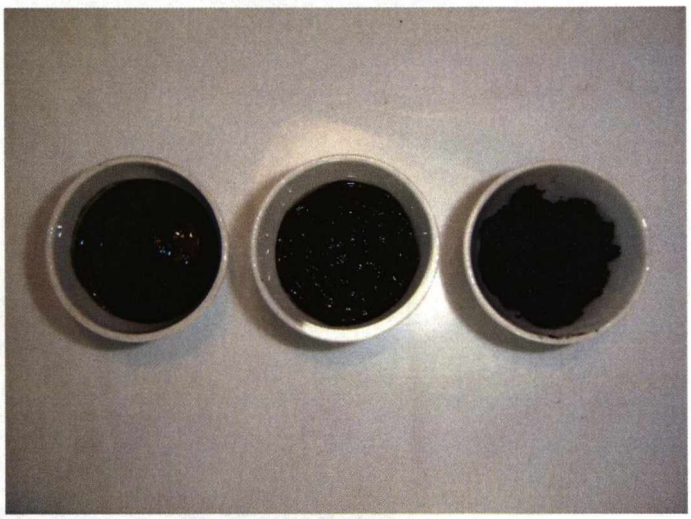


Figure 4.3 Characteristics of bottom sediments from site C in pool 6 at Brown Moss (top layer (left), middle layer (middle) and bottom layer (right))

I found a negative correlation between depth in the sediment and contents of water, loss on ignition, carbonate, Fe, TP and Ca. All variables at all sites were highest in the surface sediment and decreased gradually in the deeper layers (*Figure 4.4*). Water, loss on ignition, carbonate, Fe, TP and Ca contents appeared to be relatively high at sites A and C while carbonate content was high at site D and E. Site B had lower contents all variables measured than the other sediments.

Water content from the surface sediment in Brown Moss varied from 71.30 – 92.03 % and decreased with increasing depth. The lowest water content measured was 29.25% at 15 cm at site B. Loss on ignition in top-lying sediment ranged from 23.93 % at site B to 58.15 % at site C. The lowest loss on ignition (5.14%) was found at the lowest depth investigated. Carbonate content was highest in the top lying surface at site D (58 mg g^{-1}) and was lowest (14 mg g^{-1}) at the depth of 15 cm at site B. Maximum phosphorus content (3.31 mg P g^{-1}) was found in the surface sediment of site C while the minimum phosphorus content was 0.03 mg P g^{-1} at site B at the lowest depth. Fe content was not detectable at site B at a depth of 15 cm and was highest (4.78 mg Fe g^{-1}) in the top layer at site A. Ca content ranged from 1.22 mg Ca g^{-1} at the lowest sediment layer at site B and was highest (54.92 mg Ca g^{-1}) at the top layer at site C. Average Fe:P ratio increased with increasing depth. Fe:P ratios in the surface sediment, 1-5 cm and 5-15 cm were 1.08, 1.2 and 1.3, respectively.

Overall, multiple comparison analysis showed that phosphorus contents significantly decreased with depth but not moisture, loss on ignition and carbonate contents. Significant differences of phosphorus values between different layers of the sediment ($F = 0.8783$, $P = 0.004$) was found. Specifically, phosphorus values of the surface layer were significantly different from the bottom sediment ($P = 0.004$) but not from the middle layer ($P = 0.056$). No significant differences of moisture ($F = 2.251$, $P = 0.148$), loss on ignition ($F = 2.082$, $P = 0.167$), carbonate ($F = 3.058$, $P = 0.084$), Fe ($F = 1.159$, $P = 0.349$) and Ca ($F = 2.102$, $P = 0.165$) contents were found among surface, middle and bottom layers across all the cores.

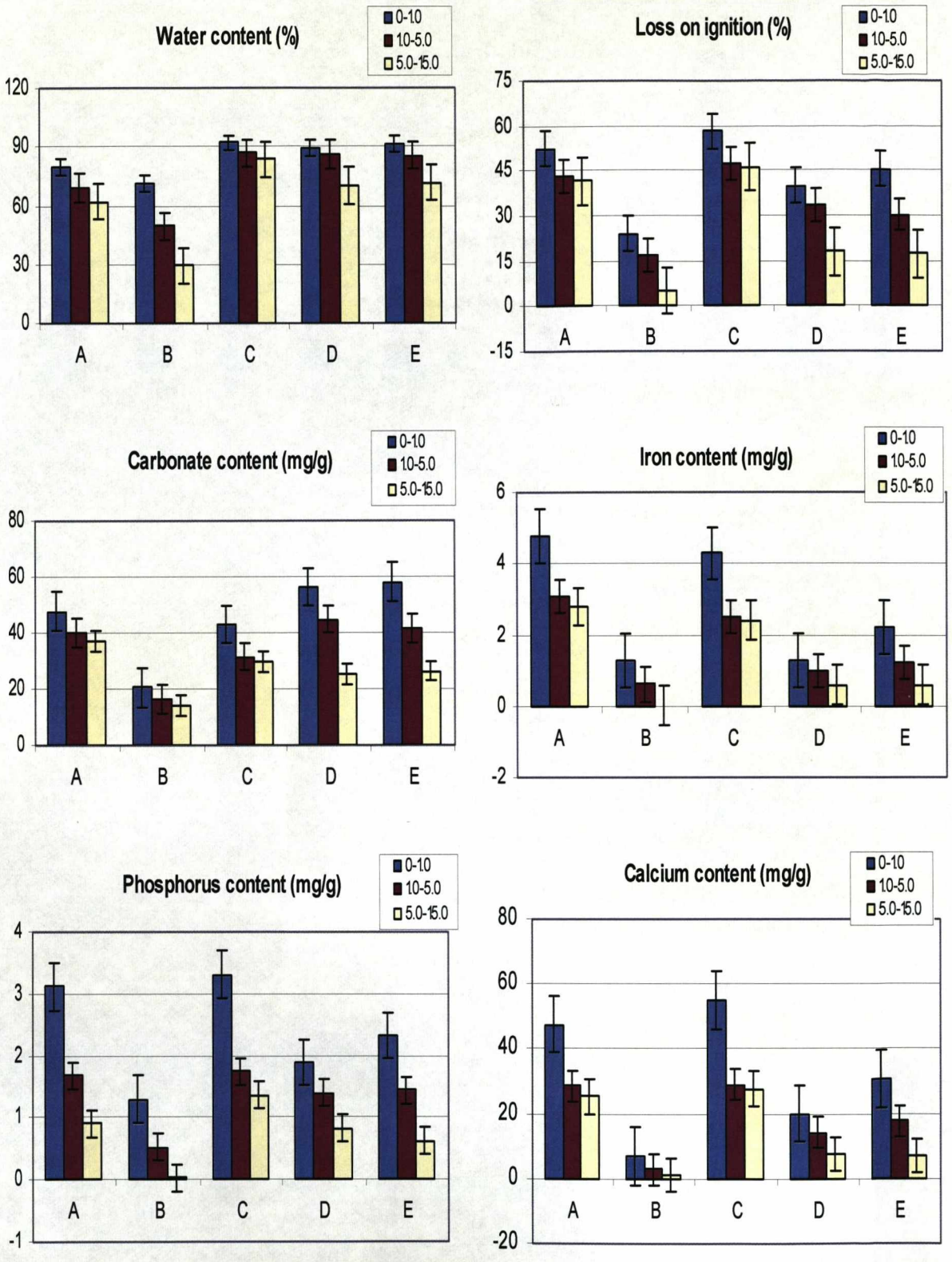


Figure 4.4 Contents of water, loss on ignition, carbonate, Fe, TP and Ca from different sediment layers in Brown Moss with standard error (n = 3)

Pearson correlation analysis among chemical variables in the surface sediment was also performed as shown in Table 4.1. There were strong positive correlations between loss on ignition and TP (r = 0.973, P = 0.005) and Ca (r = 0.975, P = 0.005) (*Figure 4.5*). Likewise, TP in the sediment was also significantly correlated with Fe (r = 0.941, P = 0.017) and Ca (r = 0.999, P < 0.001).

Table 4.1 Correlation coefficients of variables in the top-lying sediment among all cores (n = 15)

Parameters	LI	Carbonate	Fe	TP
Carbonate	0.570			
Fe	0.841	0.145		
TP	0.973**	0.396	0.941*	
Ca	0.975**	0.383	0.932*	0.999**

** Correlation is significant at the 0.01 level and

* Correlation is significant at the 0.05 level.

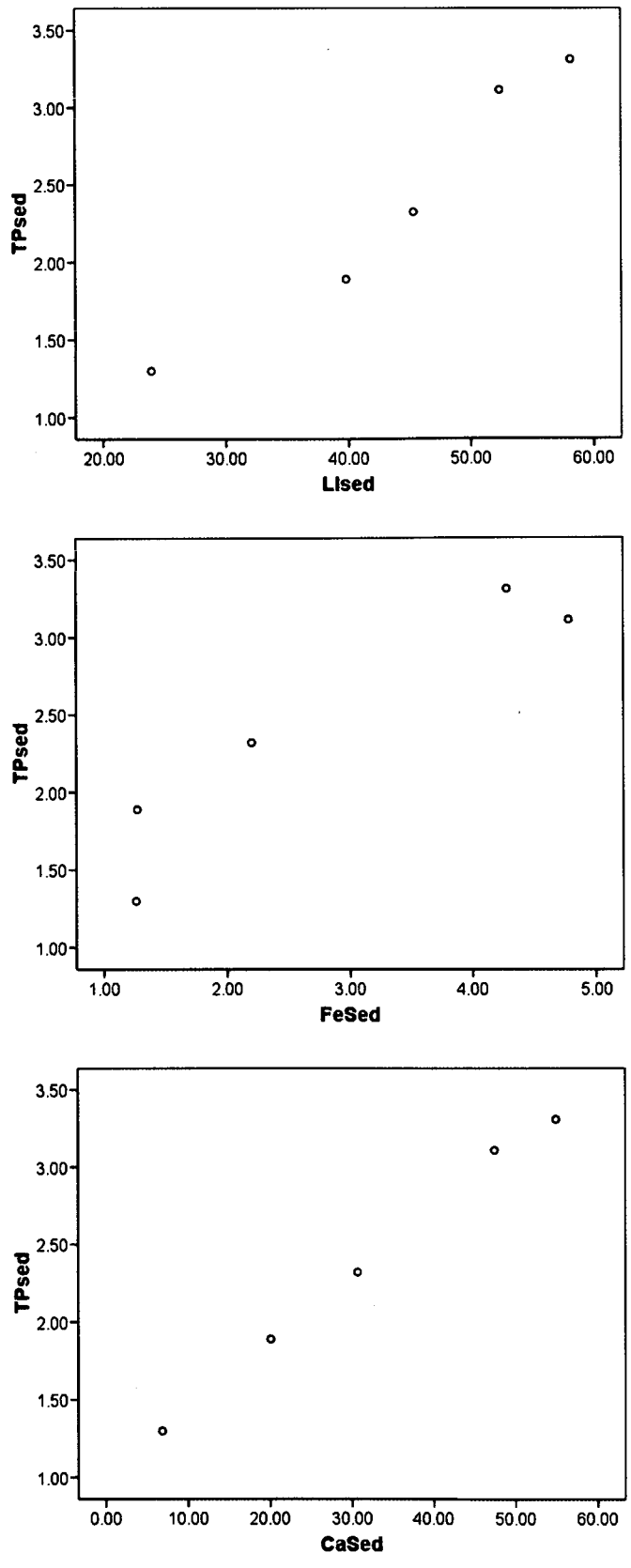


Figure 4.5 Significant relationship between TP (mg P g^{-1}) and loss on ignition (%), Fe (mg Fe g^{-1}) and Ca (mg Ca g^{-1}) contents in surface sediments (n = 5)

4.3.2 Nutrient release

Throughout study period, available net phosphate, TP, ammonium nitrogen and TN were released from the sediment but not NO_3^--N (*Figure 4.6*). Higher concentrations of nutrients appeared to be released from locations A and C whereas lower quantities of nutrients released were detected at locations B, D and E. The peak mean values of SRP and TP release were 4.3 and 7.7 mg m^{-2} day^{-1}, respectively at site A. At sites C and D, the highest release means of NH_4^+-N and TN were found with rates of 5.6 mg m^{-2} day^{-1} and 13.13 mg m^{-2} day^{-1}, respectively. The lowest mean net values of SRP and NH_4^+-N release were released from the sediment at rates of 0.5 and 0.5 mg m^{-2} day^{-1}, respectively at site E. And at site D, the lowest net release of TP was detected at a rate of 0.1 mg m^{-2} day^{-1}.

Tukey multiple comparisons revealed that values of SRP release were not significantly different among sites ($F = 1.945$, $P = 0.109$). On the contrary, significant differences of TP release rates among sites were found ($F = 3.357$, $P = 0.01$) and specifically, release rates of TP at site A were significantly different from sites B ($P_{A,B} = 0.046$) and D ($P_{A,D} = 0.020$) but not between sites A and C ($P_{A,C} = 0.691$) and sites A and E ($P_{A,E} = 0.060$). Furthermore, statistical analysis indicated no significant differences of release rates of NH_4^+-N ($F = 0.119$, $P = 0.975$), NO_3^--N ($F = 2.422$, $P = 0.053$) and TN ($F = 0.511$, $P = 0.728$) among sites.

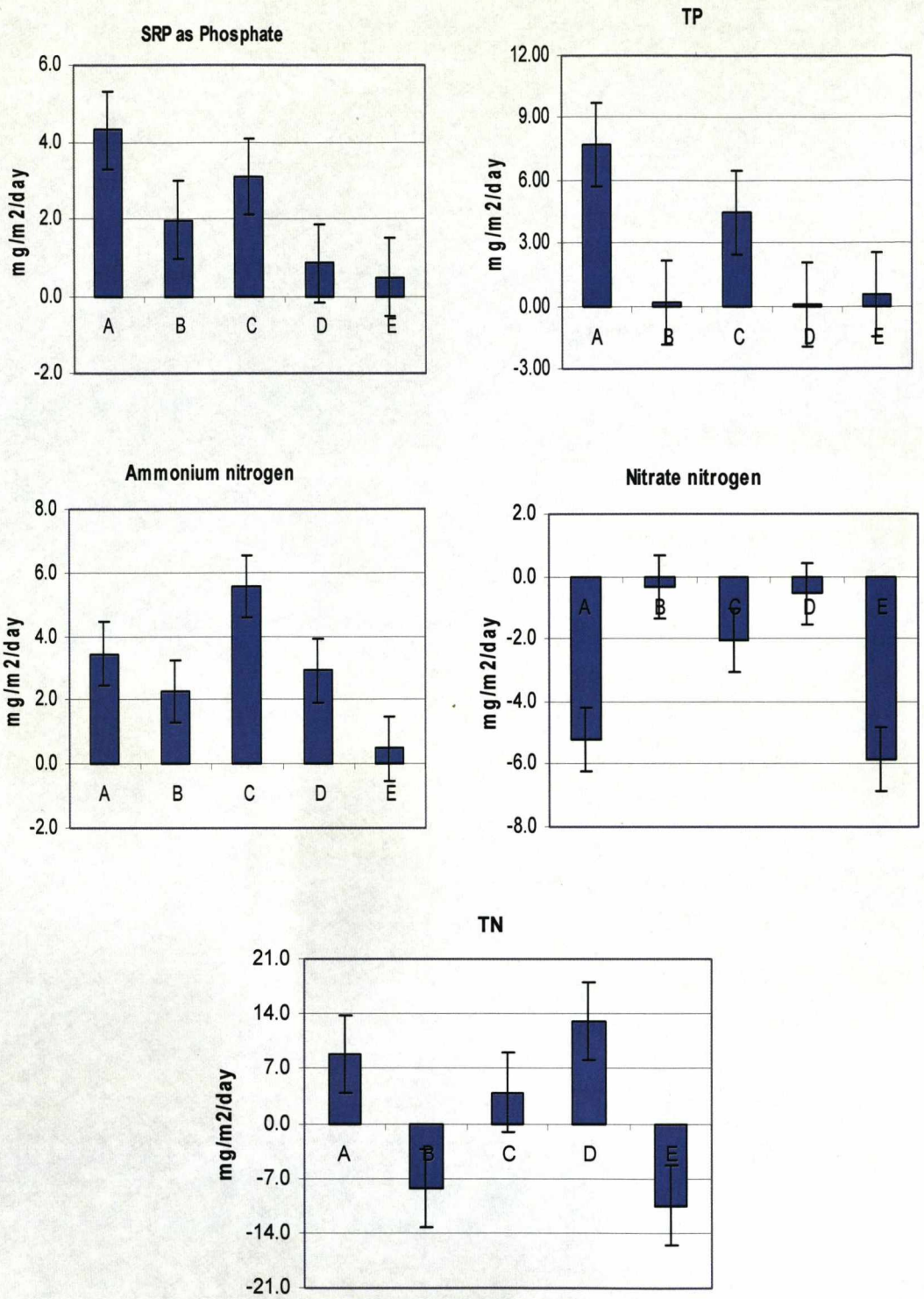


Figure 4.6 Average releases of nutrients from five locations in the main pool with standard error (n = 3)

Pearson correlation coefficients were used to determine the correlations among nutrient release and chemical variables in the surface sediments (*Figure 4.7*). I compared nutrient release and chemical variables only in the surface sediments because the surface sediments are those in contact with the water. Positive and significant correlations between TP release and Fe contents in the surface sediment was found (r = 0.957, P = 0.011). However, concentrations of TP released were not correlated with TP values in the surface sediment (r = 0.831, P = 0.081) or loss on ignition (r = 0.704, P = 0.184). Likewise, no significant correlation between NH_4^+-N release and loss on ignition in the surface sediment was found (r = 0.530, P = 0.358). See more details of correlations among variables in Table 4.2.

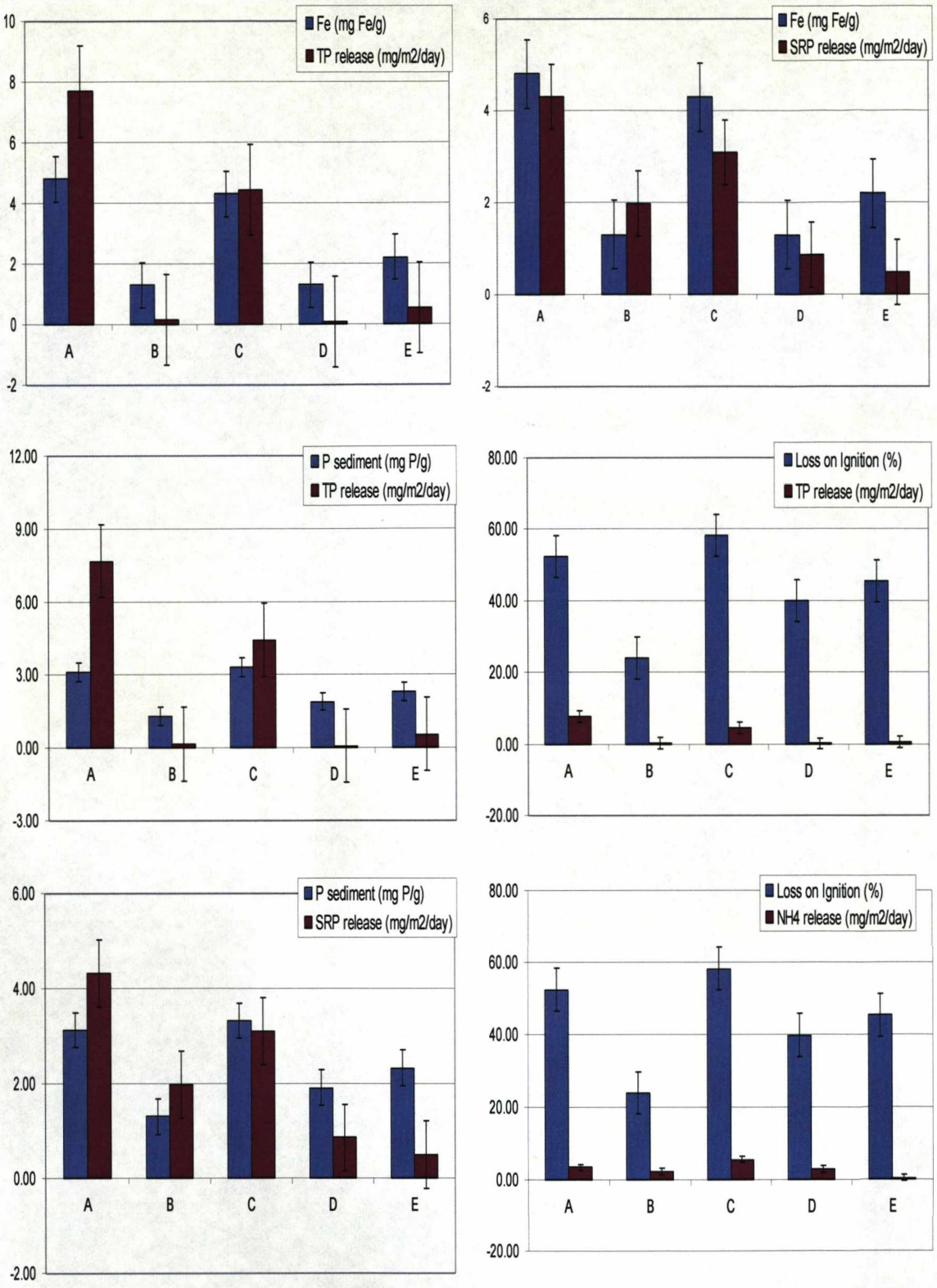


Figure 4.7 Comparison between chemical variables in the surface sediment and nutrient release with standard error
(n = 3 for sediment and n = 3 for nutrient release)

Table 4.2 Correlation coefficients of sediment chemistry and nutrient release from pool 6 at Brown Moss (n = 15)

Parameters	TP (s)	LI (s)	Fe (s)	Ca (s)	CO_3^{2-} (r)	SRP (r)	TP (r)	NH_4^+-N (r)	NO_3^--N (r)
LI (s)	0.973**								
Fe (s)	0.941*	0.841							
Ca (s)	0.999**	0.975**	0.932*						
CO_3^{2-} (s)	0.396	0.570	0.145	0.383					
SRP (r)	0.636	0.457	0.843	0.619	-0.296				
TP (r)	0.831	0.704	0.957*	0.809	0.069	-			
NH_4^+-N (r)	0.596	0.530	0.602	0.609	-0.186	0.665	0.565		
NO_3^--N (r)	-0.534	-0.532	-0.523	-0.515	-0.533	-0.172	-0.466	0.344	
TN (r)	0.379	0.415	0.324	0.358	0.365	0.373	0.432	-	-

Remark; s and r stand for sediments and release, respectively

** Correlation is significant at the 0.01 level and

* Correlation is significant at the 0.05 level.

- Auto correlated

4.4 Discussion

Because the surface sediment is in contact with the water, water content was highest in the top sediment layer and decreased gradually with increasing depth. The sediment, especially in the surface layer was uniform, fine and well mixed and this was similar to findings of Sand-Jensen (1998) stating that the sediment became coarser and more variable in grain size with increasing depth in the sediment. Deeper sediments were more heterogeneous and drier because the sediment is not in contact with water and increasing grain size is less able to absorb water. Furthermore, this could reflect diagenesis or the effects of previous drying out of the pools, which would result in oxidation of sediment.

Most sediments were black or very dark and this may be the result of chemical reactions between iron and sulphide in the sediment as described in Chapter 3. In anaerobic sediment, sulphide from sulphate reduction (SO_4^{2-}) may precipitate Fe^{2+} to form black and insoluble iron sulphide (FeS) (Mortimer 1942, 1971; Holden and Armstrong, 1986; Moss et al. 1998). Brownish colour of the sediment at site B stems from the colour of sands and gravels that are the major components.

Surface sediments contained the highest quantities of chemical variables such as organic matter and phosphorus compared with lower depths and contents of organic matter and phosphorus decreased gradually with increasing depth. Our results were consistent with a study in shallow Danish lakes done by Sondergaard et al. (1996) who reported that in most sediments a higher surface residual phosphorus was recorded as compared to depths below 20-30 cm. Meyers and Eadie (1993) also reported the same results and explained that organic carbon concentrations decreased with depth as a result of remineralisation of organic matter by microbial activity and dilution by resuspended bottom sediments.

Average amounts of total phosphorus and organic matter in sediments measured in this study were similar to other highly eutrophic/dystrophic lakes (our mean TP content and loss on ignition were 2.39 mg P g^{-1} and 43.92%, respectively). For example, in 15 Danish lakes, Jensen at al. (1992) discovered that mean content of loss on ignition in the sediment was 42% and in Lake Vallentunasjon, Sweden, a

previously sewage loaded lake, maximum total phosphorus was 1.81 mg g^{-1} DW. In high external loading lakes such as in Lake Sobygaard and Lake Sodra Bergundasjon, higher content of tot-P in the surface sediment was found at 6.1 and 6.49 mg g^{-1}, respectively (Sondergaard et al. 1993). In contrast, in mesotrophic and non-polluted lakes such as in Lake Erken, lower phosphorus content was reported ranging up to 1.23 mg g^{-1} DW (Bostrom et al. 1988).

High organic content in the top-lying sediments may potentially serve as a crucial source of available nutrients to be released into water column. Rates of P and N release were similar to other highly eutrophic lakes and comparison between the release rates of nutrients in the studied pools with other lakes can be found in the discussion section of Chapter 3. Although no significant and simple correlation between nutrient release and contents of organic matter and phosphorus in the sediment was found, phosphorus and ammonium release seems to be consistent with contents of P and organic matter in the surface sediment. In contrast, at site B where is a sandy shoreline, lower organic material in the sediments resulted in lower release of nutrients (*Figure 4.7*). This illustrates the importance of organic matter in the sediment as a nutrient source. As indicated by Bostrom and Pettersson (1982) and Phillips et al. (1994), high release of phosphorus from the sediment, due to mineralisation, is most likely to be found in the sediment where the organic phosphorus content was high. Sondergaard et al. (1999) also stated that high phosphorus concentrations due to internal loading from the sediment, with a strongly negative impact on lake water quality, are often seen in shallow lakes. For ammonium nitrogen, Dietz (1982) and Nowicki and Scott (1985) showed that average ammonium release rates over mud sediments were ten times higher than those observed on the sandy tidal flat, which was consistent with our results in that ammonium release rates were lower at sandy site B.

Experiments clearly showed that only NH_4^+-N was released from the sediment but not nitrate. This is likely due to a chemical mechanism taking place in the sediment. Under anaerobic conditions in lake sediments, nitrification does not take place due to lack of oxygen and as a result, nitrate is not released (Beutel, 2006). But ammonium nitrogen can be released from the anoxic sediment due to decomposition of organic matter (Rysgaard et al. 1994; Steingruber et al. 2001). The NH_4^+ fluxes from the

sediment can also be influenced by the activity of the benthic fauna. Blackburn and Henriksen (1983) stated that benthic animals had a considerable effect in increasing the NH_4^+ fluxes by direct excretion and by stimulation of microbial nitrogen transformations in the microenvironment around the burrow.

Besides being a source, the sediment may also be involved in controlling the nutrient release and serves as a sink and this may obscure the whole correlation analysis between nutrient release and contents of organic matter and phosphorus in the sediment in this study. Relatively high values of pH (7.4) and alkalinity (3 mEq L^{-1}) in the water at site E in particular suggested high bicarbonate and carbonate ions and this corresponded well with high content of carbonate in the sediment at site E. An important source of carbonate at site E may derive from spring at the far end of the pool and this may cause low phosphorus release at site E. Site D also had a high carbonate content and low release rate. A geological survey of Brown Moss (Environmental Consultancy, 2001) indicated that the site is thought to be underlain by an outlier of Lower Liassic mudstones, limestones, siltstones and sandstones. Limestone is a sedimentary rock composed of the mineral calcite (calcium carbonate: $CaCO_3$) (Folk, 1959; Nelson, 1978). As phosphorus can be precipitated with calcium carbonate ($CaCO_3$) in the sediment, nutrients may not be released at high rate (Koschel et al. 1983, Murphy et al. 1983; Golterman, 1988; Cooke, et al. 1993; Sondergaard et al. 2003). Jensen at al. (1998) revealed that carbonate sediments of tropical and subtropical coastal waters are thought to trap phosphorus efficiently by strong adsorption of dissolved inorganic phosphorus onto calcium carbonate ($CaCO_3$) particles. Being adsorbed onto the surface of carbonate particles probably represents the primary sink for P in carbonate sediments (Kitano et al. 1978; Jensen at al. 1998).

Fe content at site B was lowest and Fe was not detectable below 10 centimetres while content of Fe in other locations was relatively high and remained detectable in the sediment even at the depth of 15 centimetres. On average, Fe content in the surface layer of the sediment at Brown Moss was rather low (2.76 mg Fe g^{-1}) compared with other lakes. Jensen et al. (1992) measured iron content in 15 Danish lakes and found that an average of Fe in surface sediment was 40.26 mg Fe g^{-1}. Low amounts of Fe content in the sediments in the lake may be the result of the lack of surface streams/rivers that supply external Fe particles to the lake.

Although Fe content was relatively higher at sites A and C, high concentrations of phosphorus were still found to be released from the sediment. I calculated the Fe:P ratio in surface sediments of the main lake at Brown Moss and found that Fe:P was below 15. This indicates an increased chance of release of phosphorus (Jensen at al. 1992). As mentioned earlier, black, top-lying sediment suggested that the sediment in the lake was anaerobic and under anaerobic conditions, phosphorus is released. If sediment iron content is high (10 – 30 mg Fe g^{-1}, Jackson, 1989), adsorption to iron hydroxides is likely to be a major factor controlling the availability of phosphorus (Phillips et al. 1994).

In conclusion, the top surface sediment is the most important layer as maximum quantities of organic matter and phosphorus were present here and contents of all variables measured decreased with increasing depth. High phosphorus and ammonia release rates were detected at the investigated sites where high contents of phosphorus and organic matter were found. Anoxic condition in the sediment plays a crucial role in regulating nutrient release. The black sediment colour indicated an anaerobic condition in the sediment and as a consequence high release rates of phosphorus were detected. Anaerobic sediments also resulted in non release of nitrate. Furthermore, high content of carbonate in part of pool 6 caused lower release rates of phosphorus at some stations. Accordingly, the sediments in pool 6 serve not only as an important source but also as a sink of nutrients. Summary of mechanisms involved in release of nutrients from sediments in a lake is presented in Figure 4.8.

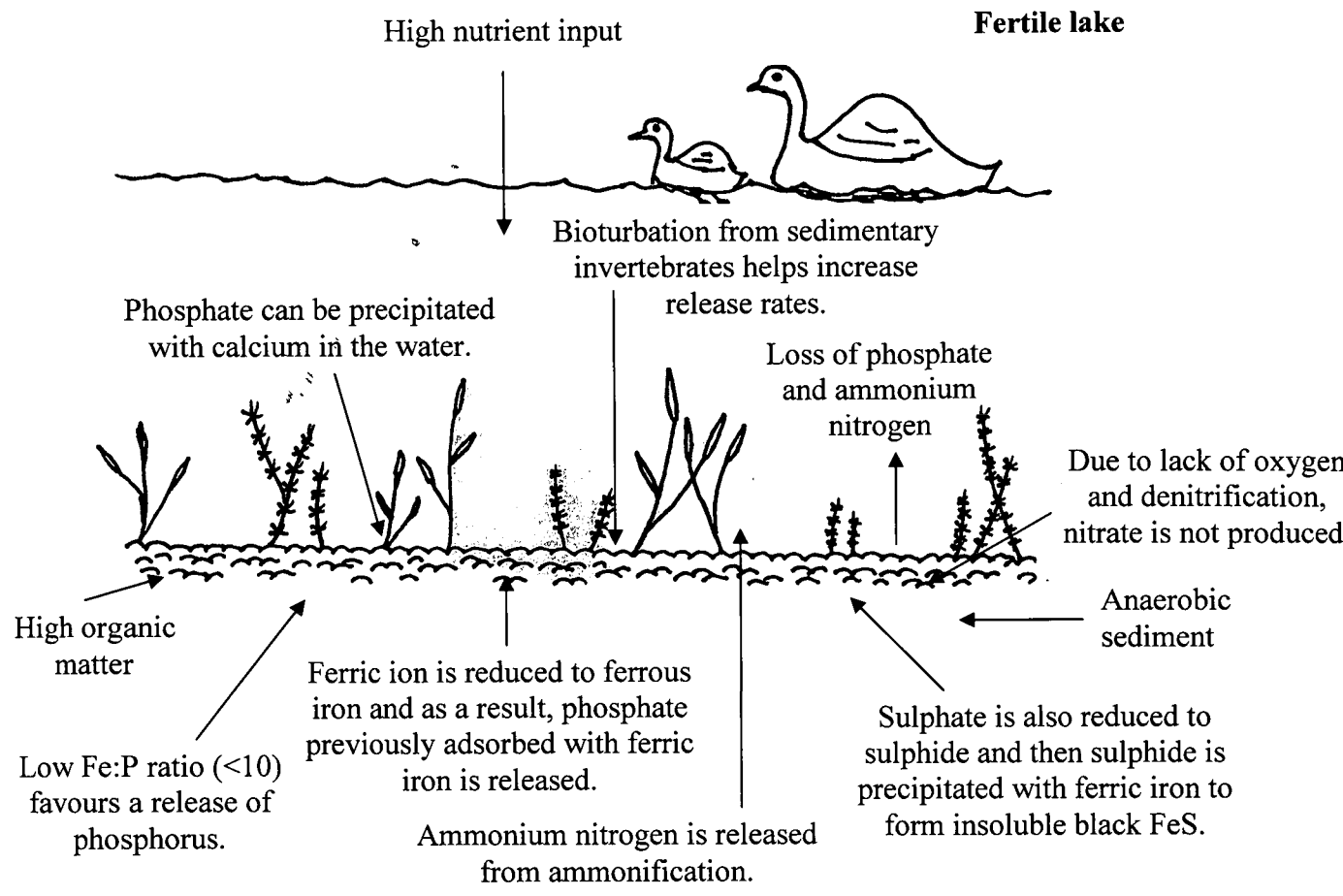


Figure 4.8 Mechanisms involved in release of nutrients from sediments in pool 6.

Chapter 5 Limnology of Brown Moss

Chapter 5
Limnology of Brown Moss

5.1 Introduction

A global survey of the size of standing waters demonstrates that most of the world's lakes are small (< 1 km^2 surface area) and shallow (< 10 m mean depth) (Wetzel, 1990). Although shallow lakes are often small systems, they are important freshwater habitats and play several roles in nature and human affairs. A shallow lake or pond is a permanent or temporary standing body of water that is sufficiently shallow to allow light penetration to the bottom sediments adequate to potentially support photosynthesis of higher aquatic plants over much of the bottom (Wetzel, 2001).

Several studies have revealed that shallow lakes and ponds even within a small geographical area are different and have their own unique limnological characteristics such as water chemistry and communities of aquatic plants and animals. For example, Irfanullah and Moss (2005) found that in Cheshire, Black Lake had low nutrient concentrations but, on the other hand, the nearby Blakemere Moss was a high nutrient lake. Therefore, limnological differences among shallow lakes and ponds may be regulated and influenced by regional and local factors.

Climate (e.g. temperature and precipitation) and landscape are regarded as important regional determinants (David et al. 2000; Shea and Chesson, 2000) that may play a crucial role in determining the nature of lakes and ponds. For examples, physical characteristics of the lakes situated in the Lake District are strongly influenced by the movement of atmospheric pressure systems across the Atlantic (George et al. 2000). In terms of regional landscape, small lakes and ponds having close contact with the adjacent terrestrial environment may be particularly affected by land use in the catchment. This is because runoff from agricultural areas may bring nutrients to ponds and may increase nutrient concentrations. Irfanullah and Moss (2005) found that the fertility of Hatchmere in Cheshire is probably a result of the land-use pattern in its catchment as the catchment bears a large number of farms. Sondergaard et al. (2005) also reported that high concentrations of nitrate in lakes likely reflect high hydraulic loading, including surface inflows from surrounding lands rich in nitrogen. In

contrast, isolated lakes and ponds in remote areas without anthropogenic influence may be less affected due to lower external nutrient input.

Apart from regional factors, local factors such as lake morphology and biological interactions of organisms such as phytoplankton, zooplankton, invertebrates, macrophytes, fish and birds may also structure lake characteristics and functions (Levine 2000; Angermeier and Winston, 1998). Lake morphology such as depth and shallowness is considered important and may be related to abundance and species richness of aquatic plants and animals. Small lakes and ponds that have a relatively greater littoral zone may allow the growth of rooted and submerged plants even at their deepest points. In contrast, in deep ponds where light can not reach the bottom, macrophytes may be scarce and less diverse. For fish, water depth is also an important factor regulating fish survival in small lakes and ponds, especially during cold winters or drought (Tonn and Magnusson, 1982).

Besides depth, lake size may be related to species richness of some organisms. Wetzel (2001) explained that biodiversity relative to lake size can be expected to be higher in small lakes and ponds where the littoral habitat heterogeneity interfaces with pelagic regions. However, some studies have shown that species richness of fish and submerged macrophytes increased markedly with lake size (Amarsasinghe and Welcomme, 2002; Bazzanti et al. 2003; Sondergaard et al. 2005).

Phytoplankton is also a key element in small lakes and ponds as a primary producer and provides the food necessary to support aquatic animal life. High densities of phytoplankton due to high nutrient loadings reflect overall biological productivity in lakes. On the contrary, shallow lakes and ponds with low concentrations of algae may be less fertile with fewer numbers of organisms. Zooplankton is also considered as important in food webs and plays a crucial role in shallow lakes and ponds. Zooplankters are not only a main food source for other animals such as invertebrate larvae and fish but they also graze on phytoplankton. Shallow lakes and ponds with high numbers and species of zooplankton may be considered productive and zooplankton may keep lakes in a clear and phytoplankton-free state compared with those with fewer numbers of phytoplankton feeders.

Because shallow lakes and ponds are small, submerged and floating macrophytes potentially cover large areas or even the whole area of them under favourable conditions (Sondergaard et al. 2005). Presence and abundance of macrophytes may promote and support invertebrate life since aquatic vegetation serves as food sources and shelters for such animals. The community of invertebrates associated with macrophytes is varied, with Chironomidae, Oligochaeta and Mollusca often most abundant (Soszka, 1975). In contrast, less abundant plant species in lakes and ponds may result in low diversity of macro invertebrate assemblages. A study by Timms and Moss (1984) and Burks et al. (2001) showed that macrophytes can increase survival of invertebrates such as *Daphnia* populations from fish predation in shallow lakes.

In shallow lakes and ponds, fish are top predators and are potentially important regulators of other organisms. They may structure zooplankton and invertebrate communities and may also influence aquatic macrophytes. In the presence of fish, large zooplankton such as *Daphnia* may disappear from lakes because fish feed on them (Moss et al. 1997). Furthermore, smaller zooplankton such as *Cyclops* and rotifers that are ignored by fish or can escape from fish attack may become abundant and dominant. A study done by Galbraith (1967) showed that rainbow trout and yellow perch were very size selective and usually consumed only *Daphnia* over 1.3 mm in size while ignoring the many and often more numerous smaller zooplankters. In contrast, in the absence of fish due to winter fish kill or drying out, small lakes and ponds may have a more diverse community of zooplankton and large cladocerans (Sondergaard et al. 2005). Also, other invertebrate predators may become important and take over the role of fish (Hobek et al. 2002).

Since herbivorous fish may feed on submerged macrophytes, abundance of fish in shallow lakes and ponds may be likely to damage plant communities. Several studies have confirmed that fish such as the common carp (*Cyprinus carpio* (Linnaeus)) is associated with loss of plants in shallow lakes (Crivelli, 1983; Fletcher et al. 1985).

Migratory and residential aquatic birds living in lakes may also have an impact. As many lakes and ponds are natural habitats for waterfowl, gatherings of such birds may influence lake ecosystems. First, because macrophytes are the main diet of birds such as coots and ducks (Moss et al. 1997), waterfowl may destroy plant communities.

Experiments in Lake Stigsholmboth, Denmark revealed a number of significant differences between macrophyte growth in protected and unprotected areas and this suggested a negative effect of waterfowl grazing (Sondergaard et al. 1996). Also birds may enrich pools through their dropping as shown in New Mexico by Brandvold et al. (1976) where birds increased nitrogen and phosphorus downstream from where they were roosting. Lastly, some species of waterfowl feed on fish and thus birds may potentially affect food web structure in shallow lakes and ponds. Draulands (1988) showed that herons consumed a maximum of 8% of the total fish yield in one lake.

The internal sediment may also play a major role in nutrient cycling in isolated lakes and ponds, especially those having a relatively low water volume and low external input (Tessier and Woodruff, 2002). Bennion and Smith (2000) found high inter-annual variability in phosphorus which tended to be highest in the most enriched water and this possibly reflected the high impact by the sediment on seasonal nutrient concentrations. Sondergaard et al. (2005) also stated that the importance of internal processes is high in small lakes as these usually have no surface outflows and much of the phosphorus entering the lakes will be retained and potentially recycled within the lake.

At Brown Moss, there are several shallow pools of different size, depth and features that may result in different chemistry and ecological characteristics (*Figure 5.1*). Particular limnological features may explain the present trophic states of them and may determine the systems in each pool. Therefore, first I investigated current water chemistry and biological variables in particular pools to try to understand the complexity in them and their seasonal variation.

Figure 5.1 Pools at Brown Moss

Although the pools are located close to one another, their environmental and limnological features may be different as a result of local factors such as presence and absence of birds and fish. Equally, there may be similarities caused by regional influence such as types of land use and climate. I determined the influence of a variety of factors on abiotic and biotic characteristics of the pools to distinguish similarity and dissimilarity among them and to establish what factors are most crucial. I also tested the hypothesis that conditions in adjacent pools are determined more by local conditions than regional environmental factors and vice versa.

5.2 Study pools

The studied sites in this research are located in Brown Moss, a Site of Special Scientific Interest (SSSI), where is situated at grid reference SJ5639, to the south-east of Whitchurch, North Shropshire District, Shropshire and is managed by Shropshire County Council. The nature reserve consists of heathland, woodland and a series of shallow pools which often dry out and with considerable differences, resulting in a tremendous range of wild plants and animals (Jones, 1993). I chose three pools for comparative studies since they are different in morphology, location and limnological characteristics (*Figure 5.2*).

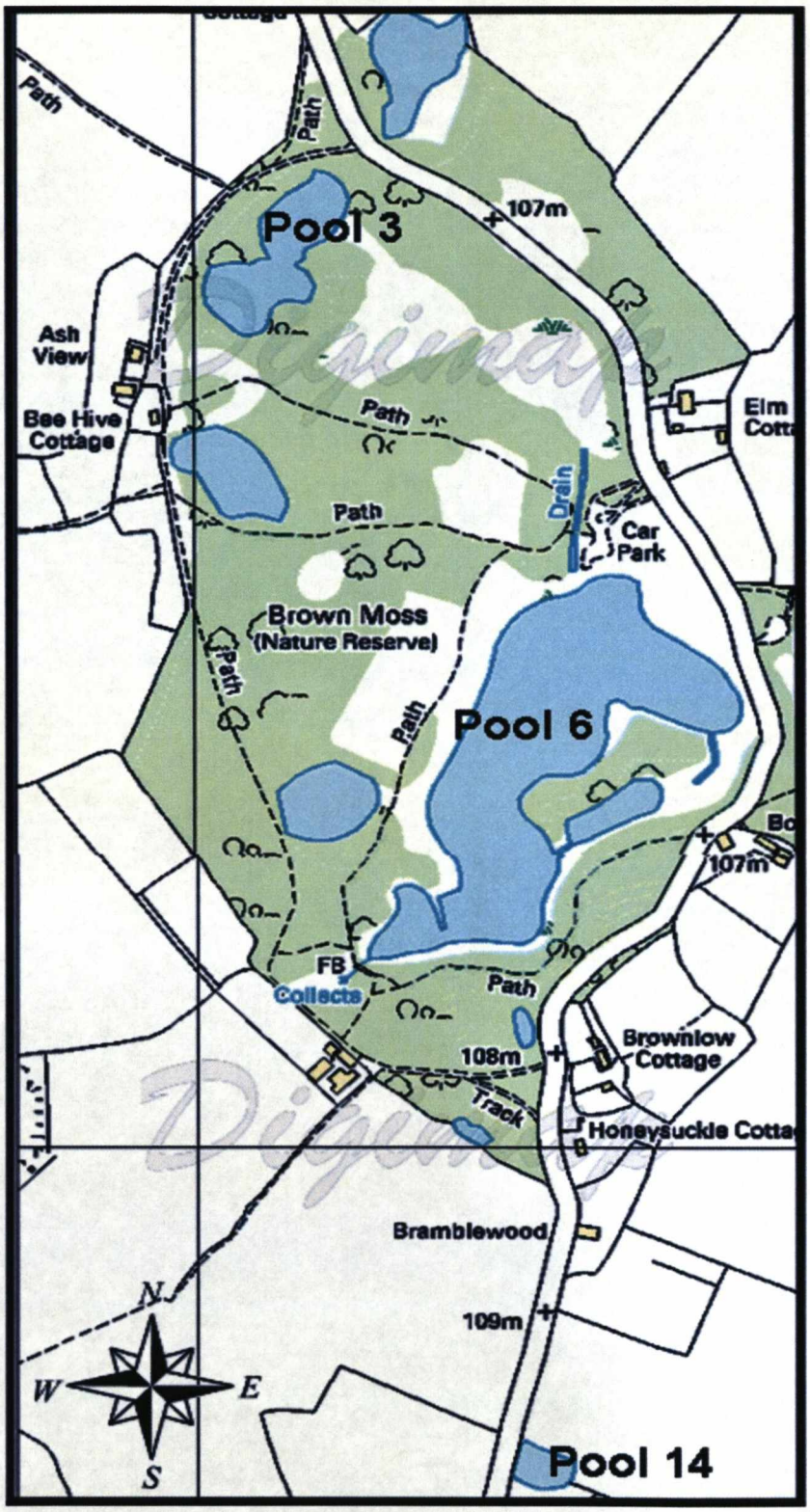


Figure 5.2 Study pools 3, 6 and 14

Pool 3 is located to the north of Brown Moss (*Figure 5.3*). Water surface area of the pool is 2,700 m^2 and mean depth is 27±15 cm. The pool is surrounded by mature woodland and the margins of the pool are encroached by a species-rich fringe. The scrub around the pool has been removed from the immediate vicinity (Lockton and Whild, 2003). This pool is relatively shallow, small in size and fishless.

Pool 6 is the biggest pool at Brown Moss (*Figure 5.3*). This pool is shallow and is a natural habitat of residential and migratory aquatic birds. Along the north-east side, close to the car park, there is a sandy shoreline that has been maintained by frequent use by people and by grazing by waterfowl, which are fed in this area by visitors (Lockton and Whild, 2003). There are no direct natural surface inflows and outflows and therefore, the pool receives water mostly from rain through soil in the catchment area and it is likely that water levels in the pool are linked to fluctuations of rainfall and evaporation. Marginal trees around the pool have been cut down and the pool has been dredged. In some years, this pool has dried out and all fish were removed from the pool in 1987. Water surface area of the pool is 29,700 m^2 and mean depth is 31±16 cm.

Pool 14 is situated outside the Brown Moss SSSI boundary and is owned by a private land owner (*Figure 5.3*). This pool is relatively small but deeper than pools 3 and 6. The pool is surrounded by intensive agricultural areas and grazed pastures and is a fish habitat. Fish such as carp (*Cyprus carpio*) are reported to be found in the pool. Water surface area of the pool is 1,200 m^2 and mean depth is 54±15 cm.

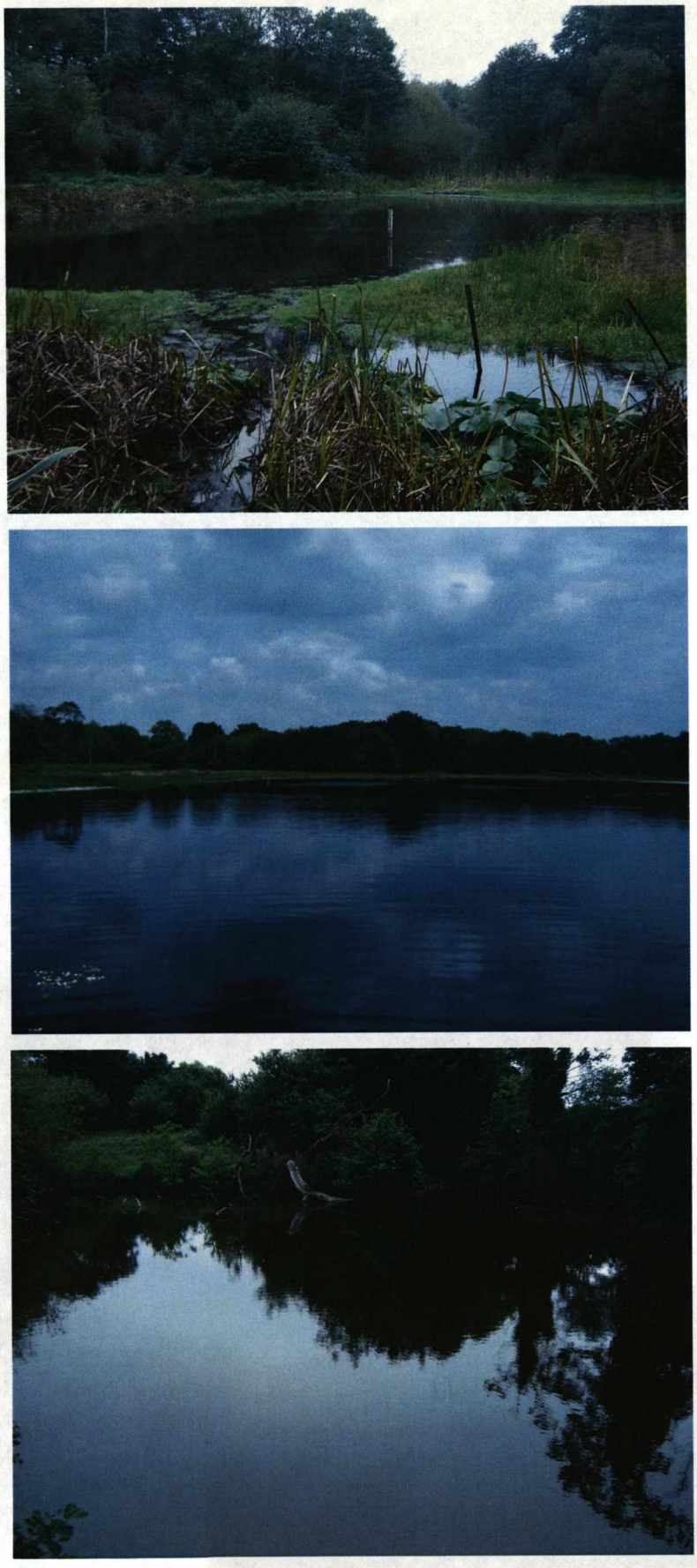


Figure 5.3 Characteristics of the pools 3, 6 and 14 (from top to bottom)

5.3 Methods

5.3.1 Physico-chemical properties

Methods for physico-chemical variables are given in Chapter 2, section 2.3.

5.3.2 Biological variables

5.3.2.1 Phytoplankton

Methods are given in Chapter 2, section 2.3.

5.3.2.2 Zooplankton

Zooplankton samples were collected fortnightly from the studied lakes from April 2006 to October 2006. I passed 10 litres of water from the studied lakes through a 63 μm mesh sieve and subsequently preserved samples with 70% ethanol in plastic bottles. Samples were identified to genus level and counted using a Wild M40 inverted microscope.

5.3.2.3 Macro invertebrates

Sedimentary invertebrates were sampled from the pools using an Ekman grab (15cm×15cm×20cm) (five samples per site) once a month from November 2004 to October 2006. Samples were transferred to wide-mouthed plastic bottles and preserved with 70% ethanol. All samples were sieved through a 335-μm mesh sieve and then were sorted from spreads on a white tray. All benthos taxa were identified up to species level. Invertebrates associated with the plants were also studied using a 250 μm mesh net swept for one minute through the plant bed in each site where there was such a bed. Animals were transferred to wide-mouthed plastic bottles and preserved with 70% ethanol. Then, samples were sorted, identified to up to species level and counted. After being identified and counted, all macroinvertebrates were dried at 105 ^{o}C for 24 hours and weighed together (Balint et al. 1998) to obtain biomass values.

5.3.2.4 Macrophytes

Surveys of macrophytes in the studied pools were carried out in the summers (July and August) of 2004-2006. I surveyed vegetation along transect across the studied pools using a 1m×1m quadrat placed at 5-m intervals. Plants found in a quadrat were identified to species level. Biomass of vegetation was estimated using percentage volume infestation values (PVI values) obtained from areas of cover and height of plants above the bottom. Photographs of plants present in pools in each summer were also taken.

5.3.2.5 Aquatic birds

Waterfowl in pools 3, 6 and 14 were studied and counted by species every two weeks using binoculars (10×50 wide angle). I counted water birds around 13.30 pm and did not disturb birds while counting. The identification of waterfowl was based on expert help and bird guide books (Johnson, 2003; Flegg and Hosking, 1990).

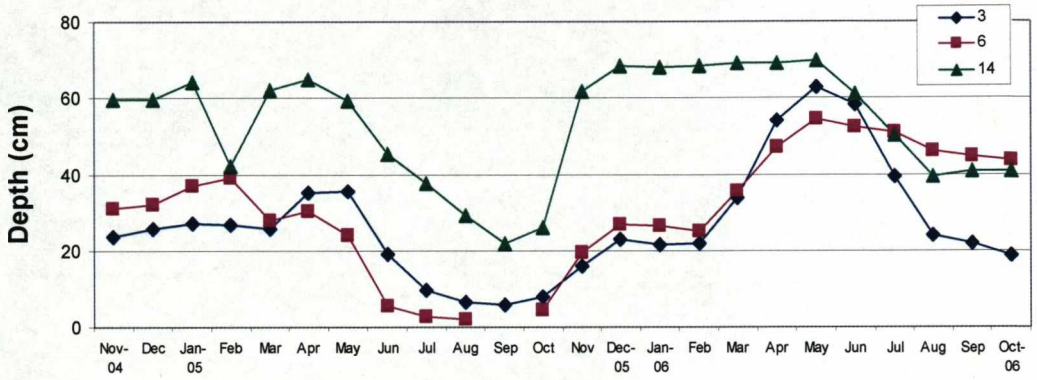
5.3.3 Statistical analysis

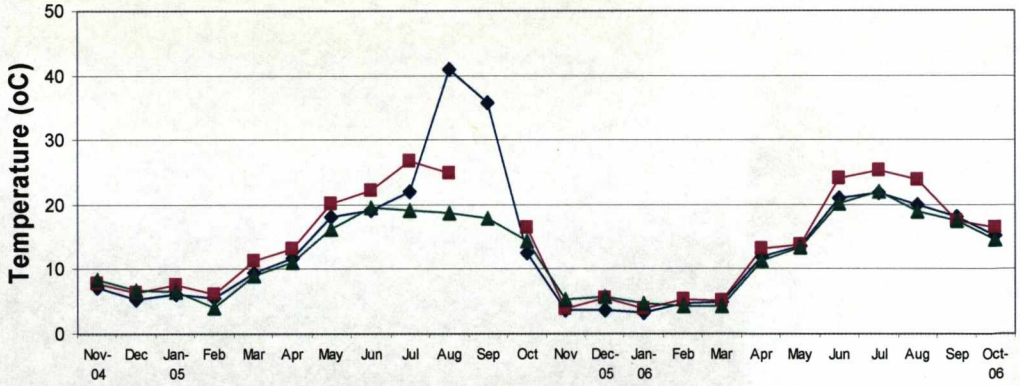
Statistical analyses were done by SPSS 14.0 for Windows. Pearson correlation coefficients (r) were used to determine correlation among variables and to test whether variables among pools act differently or not. I also applied Tukey honestly significant difference (HSD) analysis to test for significant differences between pairs of means of abiotic and biotic variables. PRIMER 6.0 was also used to indicate similarity and difference of variables among the studied pools.

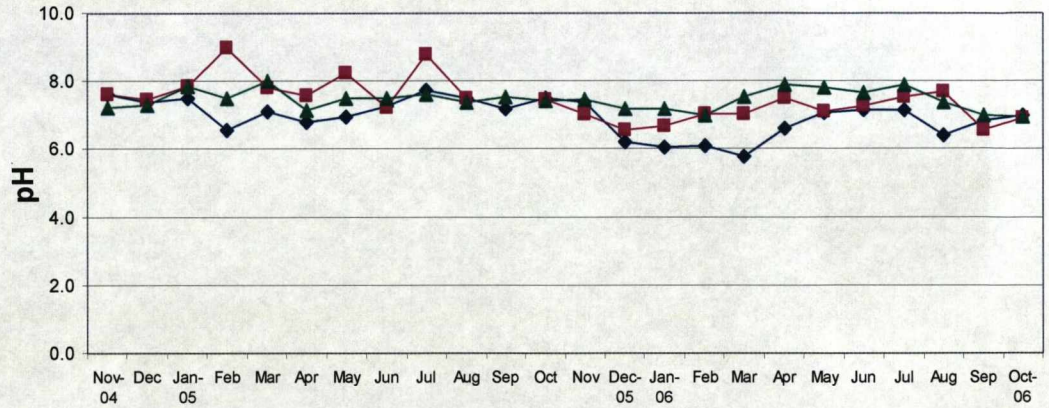
5.4 Results

5.4.1 Physico-chemical properties

Physico-chemical data have already been presented in the context of the entire system in Chapter 2 but the data are shown again for convenience of biological variables in Figures 5.4-5.6.







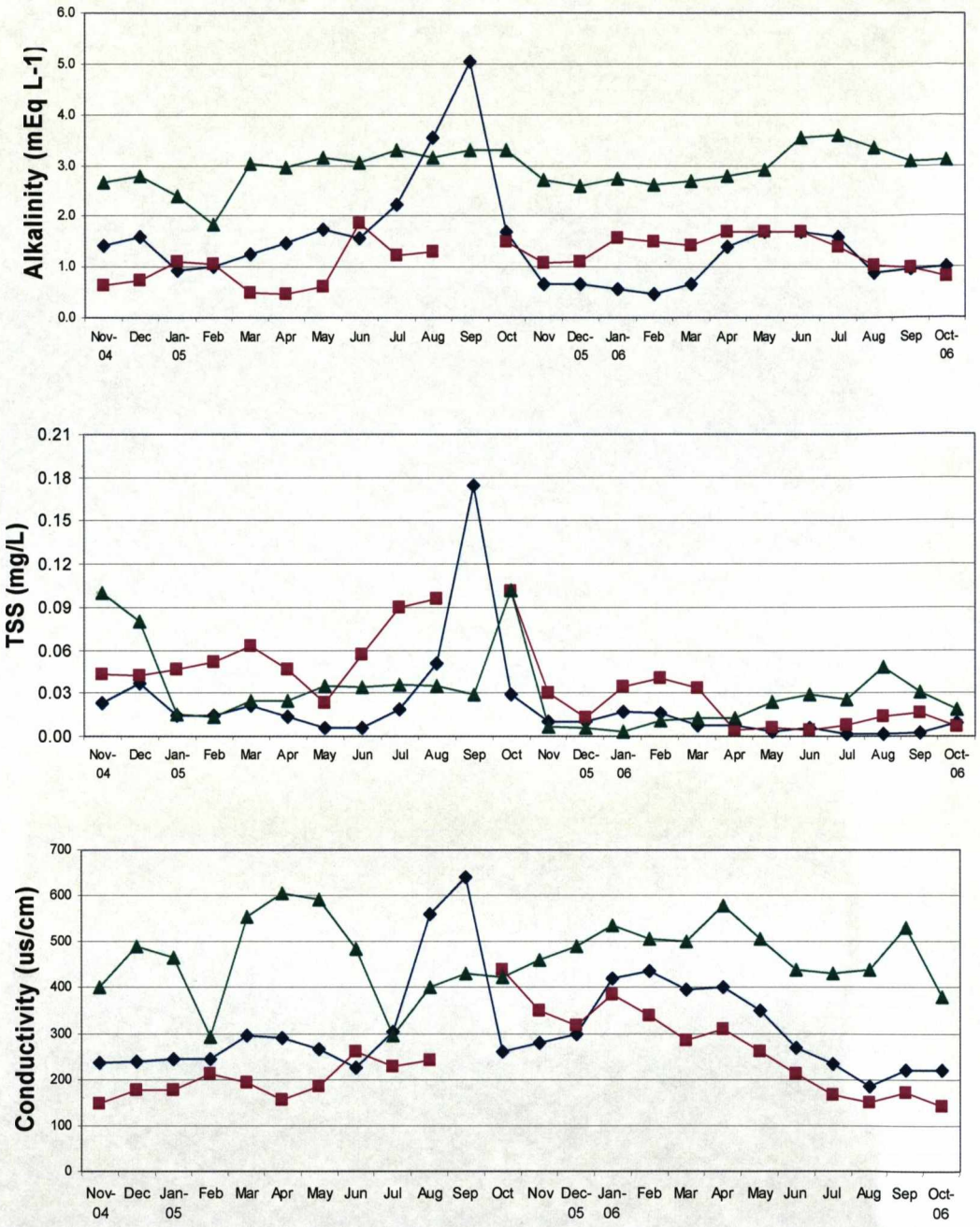


Figure 5.4 Seasonal dynamics of physical and chemical variables of pools 3, 6 and 14

I calculated correlation coefficients for physical and chemical variables among pools 3, 6 and 14 and found that seasonal variations of depth and temperatures in pool 3 significantly correlated with that in pools 6 and 14 (*Table 5.1*). Furthermore, significant correlations of conductivity and TSS between pools 3 and 6 were found and alkalinity of pool 3 was significantly correlated with that in pool 6. However, no significant correlation between pH among pools was found.

Table 5.1 Correlation coefficients of physical and chemical variables among pools 3, 6 and 14 (n = 49) (continued)

Variables	Depth 3	Depth 6	Depth 14	Temp 3	Temp 6	Temp 14	pH 3	pH 6
Depth 6	0.757**							
Depth 14	0.639**	0.433*						
Temp 3	-0.201	-0.192	-0.758**					
Temp 6	0.023	-0.092	-0.692**	0.897**				
Temp 14	0.000	-0.009	-0.680	0.839**	0.977**			
pH 3	-0.213	-0.320	-0.461	0.402	0.434*	0.449*		
pH 6	-0.067	-0.189	-0.163	0.173	0.251	0.103	0.372	
pH 14	0.448*	-0.191	0.162	0.121	0.202	0.151	0.256	0.388

** Correlation is significant at the 0.01 level and

* Correlation is significant at the 0.05 level.

Table 5.1 Correlation coefficients of physical and chemical variables among pools 3, 6 and 14 (n = 49)

Variables	Alk 3	Alk 6	Alk 14	TSS 3	TSS 6	TSS 14	Conduct 3	Conduct 6
Alk 6	0.052							
Alk 14	0.430*	0.114						
TSS 3	0.831**	-0.106	0.144					
TSS 6	0.517*	-0.057	-0.024	0.702**				
TSS 14	0.235	-0.314	0.304	0.131	0.395			
Conduct 3	0.636**	0.361	0.027	0.713**	0.247	-0.275		
Conduct 6	-0.162	0.591**	-0.153	0.205	0.220	-0.168	0.479*	
Conduct 14	-0.223	-0.088	0.053	-0.156	-0.349	-0.206	0.066	0.102

** Correlation is significant at the 0.01 level and

* Correlation is significant at the 0.05 level.

Tukey analysis was used to compare means of all variables and it revealed significant differences of depth between pools 3 and 14 ($P_{3,14} < 0.001$) and between pools 6 and 14 ($P_{6,14} < 0.001$) (*Figure 5.5*). But no significant difference of depth between pools 3 and 6 was found ($P_{3,6} = 0.658$). Furthermore, temperature ($P_{3,6} = 0.998$, $P_{3,14} = 0.716$ and $P_{6,14} = 0.753$) and TSS ($P_{3,6} = 0.169$, $P_{3,14} = 0.407$ and $P_{6,14} = 0.845$) between pools 3, 6 and 14 did not differ significantly. But pH values between pools 3 and 6 ($P_{3,6} = 0.004$) and between pools 3 and 14 ($P_{3,14} = 0.003$) were significantly different. Our results also indicated no significant difference of pH values between pools 6 and 14 ($P_{6,14} = 0.997$). Alkalinity ($P_{14,3} < 0.001$, $P_{14,6} < 0.001$) and conductivity ($P_{14,3} < 0.001$, $P_{14,6} < 0.001$) of pool 14 significantly differed from pools 3 and 6, and significant difference of conductivity between pools 3 and 6 was also found ($P_{3,6} = 0.024$). Lastly, alkalinity in pool 3 did not significantly differ from that in pool 6 ($P_{3,6} = 0.221$).

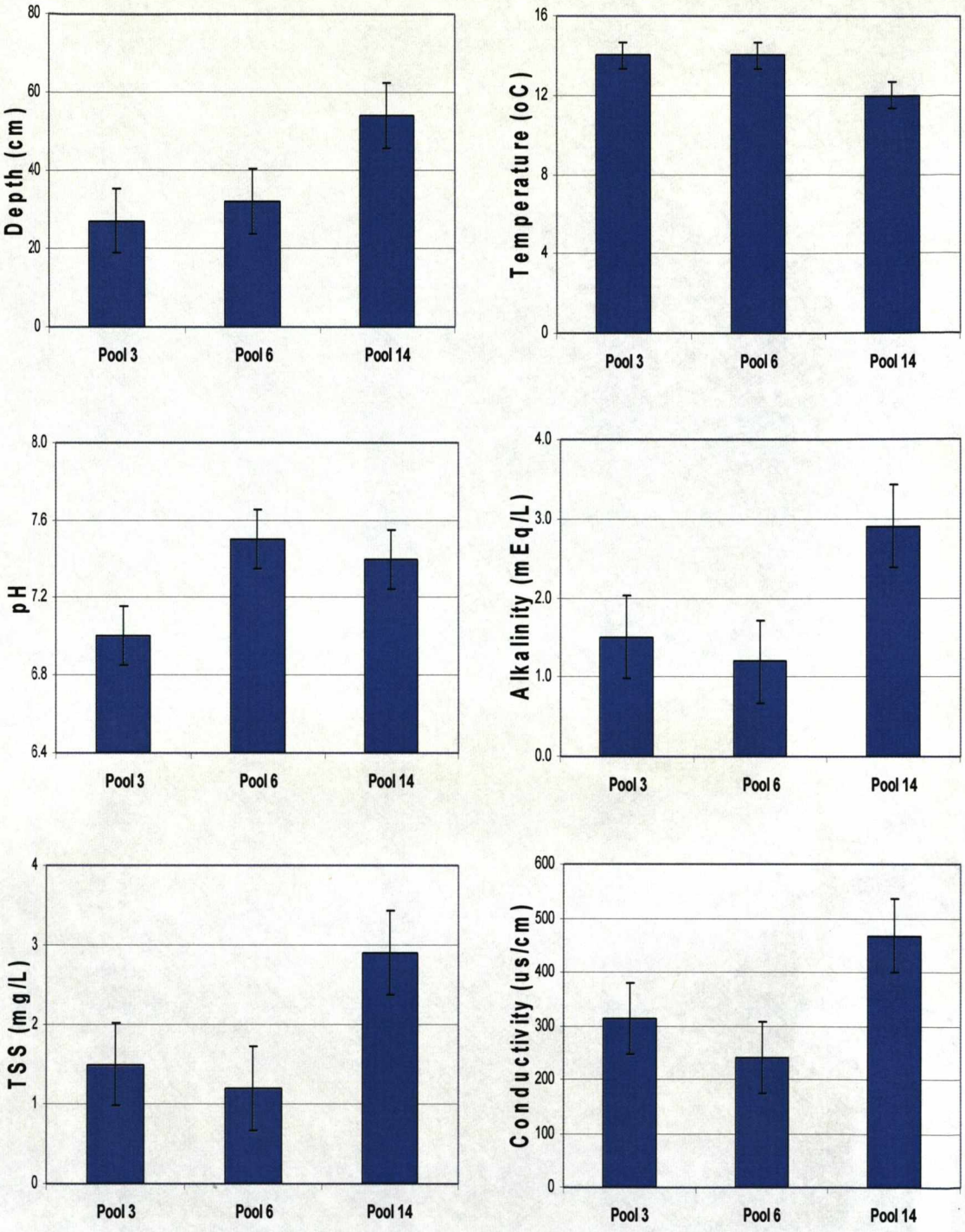


Figure 5.5 Comparison of physical and chemical variables in pools 3, 6 and 14 with standard error (n = 49)

Concentrations of nutrients in pools 3, 6 and 14 changed seasonally and showed similar trends (*Figure 5.6*). SRP and TP concentrations in all studied pools tended to increase in spring onward and were highest in summer. SRP and TP values were rather higher in pools 6 and 14 and were lower in pool 3 (*Figure 5.5*). Average concentrations of SRP in pools 3, 6 and 14 were 41±65 μg L^{-1}, 104±140 μg L^{-1} and 92±93 μg L^{-1}, respectively and average TP concentrations in pools 3, 6 and 14 were 172±121 μg L^{-1}, 389±254 μg L^{-1} and 351±186 μg L^{-1}, respectively. NO_3^--N concentrations increased in winter and were lower in other seasons. In pool 14, concentrations of NO_3^--N increased dramatically between November to February when water levels started to increase. NO_3^--N concentrations were highest in pool 14 (1.27±1.63 mg L^{-1}) and lower in pool 6 (0.18±0.27 mg L^{-1}) and 3 (0.04±0.06 mg L^{-1}), respectively.

NH_4^+-N concentrations increased twice a year in winter and in summer. Especially in pools 3 and 6, concentrations of NH_4^+-N increased substantially in summer 2005. NH_4^+-N values were highest in pool 6 and were lower in pools 3 and 14. Average concentrations of NH_4^+-N in pools 3, 6 and 14 were 189±308 μg L^{-1}, 773±1300 μg L^{-1} and 183±234 μg L^{-1}, respectively. TN in pools 3 and 6 slightly increased in winter and remained high in late summer to autumn. In pool 14, TN concentrations were high in winter and were lower in other seasons. Average concentrations of TN were highest in pools 6 (4±2.4 mg L^{-1}) and 14 (4±1.4 mg L^{-1}) and lower in pool 3 (2±1.2 mg L^{-1}).

Analysis of correlation coefficients revealed that seasonal concentrations of SRP, NH_4^+-N and TN of pool 3 significantly correlated with those in pool 6 (*Table 5.2*) but not with pool 14. Significant correlation of SRP concentrations between pool 6 and 14 was also found. In contrast, I found no significant correlation of concentrations of TP and NO_3^--N among the pools.

Table 5.2 Correlation coefficients of nutrients among pools 3, 6 and 14 (n = 49) (continued)

Variables	SRP 3	SRP 6	SRP 14	TP 3	TP 6	TP 14	NH_4^+-N 3	NH_4^+-N 6
SRP 6	0.577**							
SRP 14	0.276	0.548**						
TP 3	-	0.236	0.056					
TP 6	0.495*	-	0.140	0.328				
TP 14	0.450*	0.441*	-	0.301	0.126			
NH_4^+-N 3	0.730**	0.521*	0.245	0.618**	0.528**	0.554*		
NH_4^+-N 6	0.558**	0.746**	0.257	0.474*	0.274**	0.417*	0.767**	
NH_4^+-N 14	-0.134	0.073	0.312	-0.170	-0.124	0.036	0.037	0.036

** Correlation is significant at the 0.01 level and

* Correlation is significant at the 0.05 level.

- autocorrelated

Table 5.2 Correlation coefficients of nutrients among pools 3, 6 and 14 (n = 49)

Variables	NO_3^--N 3	NO_3^--N 6	NO_3^--N 14	TN 3	TN 6
NO_3^--N 6	0.128				
NO_3^--N 14	0.233	0.372			
TN 3	0.133	0.137	-0.403		
TN 6	0.100	0.068	-0.158	0.470*	
TN 14	0.114	0.205	0.915**	-0.372	0.606

** Correlation is significant at the 0.01 level and

* Correlation is significant at the 0.05 level.

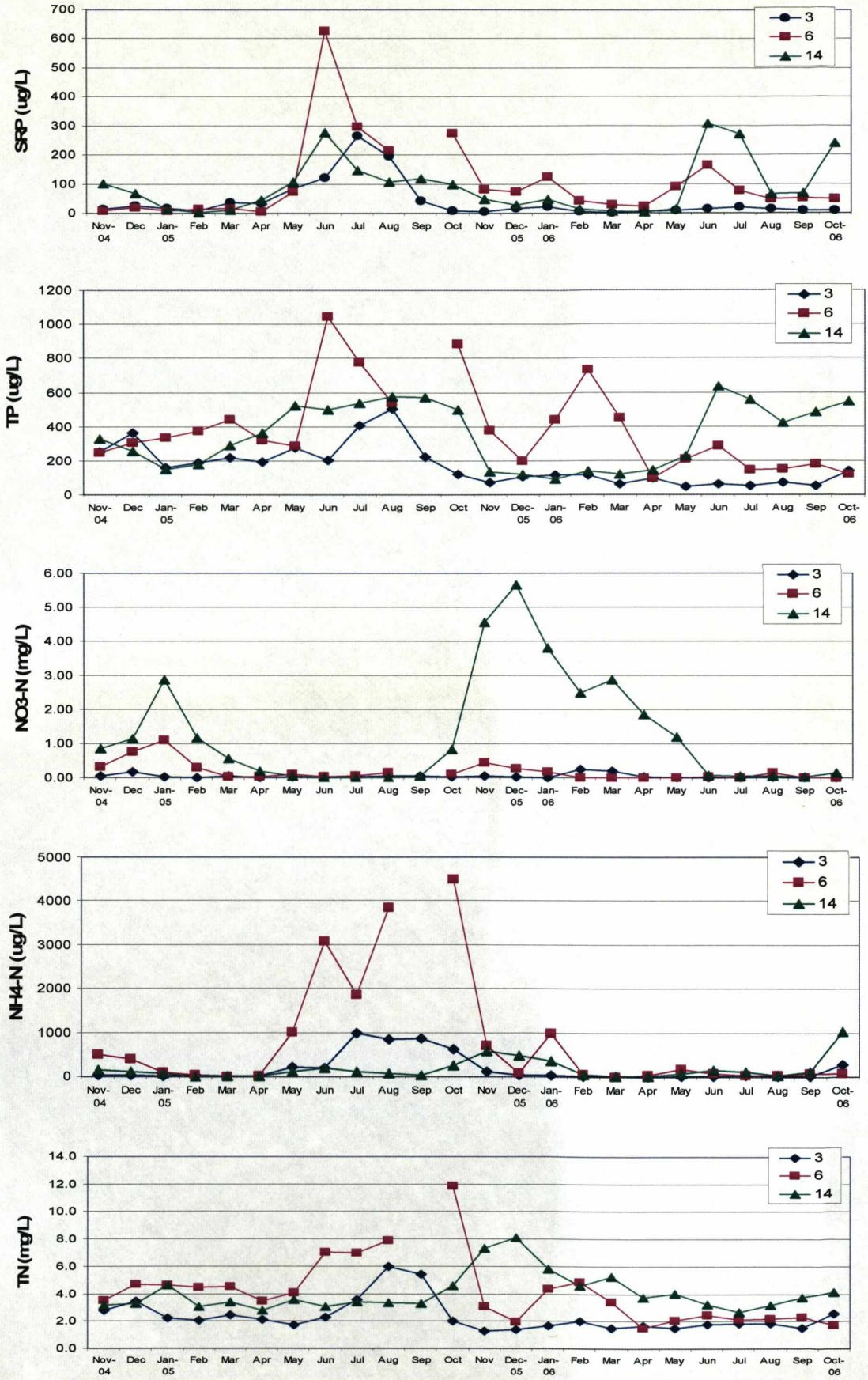


Figure 5.6 Seasonal dynamics of chemical variables of pools 3, 6 and 14

Tukey HSD analysis showed no significant differences of SRP concentrations among pools 3, 6 and 14 ($P_{3,6}$ = 0.096, $P_{3,14}$ = 0.208 and $P_{6,14}$ =0.909) (*Figure 5.7*). TP concentrations in pool 3 significantly differed from pools 6 and 14 ($P_{3,6}$ <0.001, $P_{3,14}$ = 0.006). However, statistical analysis indicated no significant difference of TN between pools 6 and 14 ($P_{6,14}$ = 0.777).

Significant differences in concentrations of NO_3^--N between pools 14 and 3, and between pools 14 and 6 ($P_{14,3}$ < 0.001 and $P_{14,6}$ < 0.001) were found. However, concentrations of NO_3^--N in pool 3 were not significantly different from that in pool 6 ($P_{3,6}$ = 0.881). In addition, significant differences of NH_4^+-N concentrations between pools 6 and 3, and between pools 6 and 14 ($P_{6,3}$ = 0.031 and $P_{6,14}$ = 0.029) were found. In contrast, NH_4^+-N concentrations of pool 3 did not differ from those in pool 14 ($P_{3,14}$ = 1.00). I also found significant differences of TN concentrations between pools 3 and 14 ($P_{3,6}$ = 0.002) and between 3 and 6 ($P_{3,14}$ = 0.003) and values of TN between pools 6 and 14 did not significantly differ ($P_{6,14}$ = 0.029).

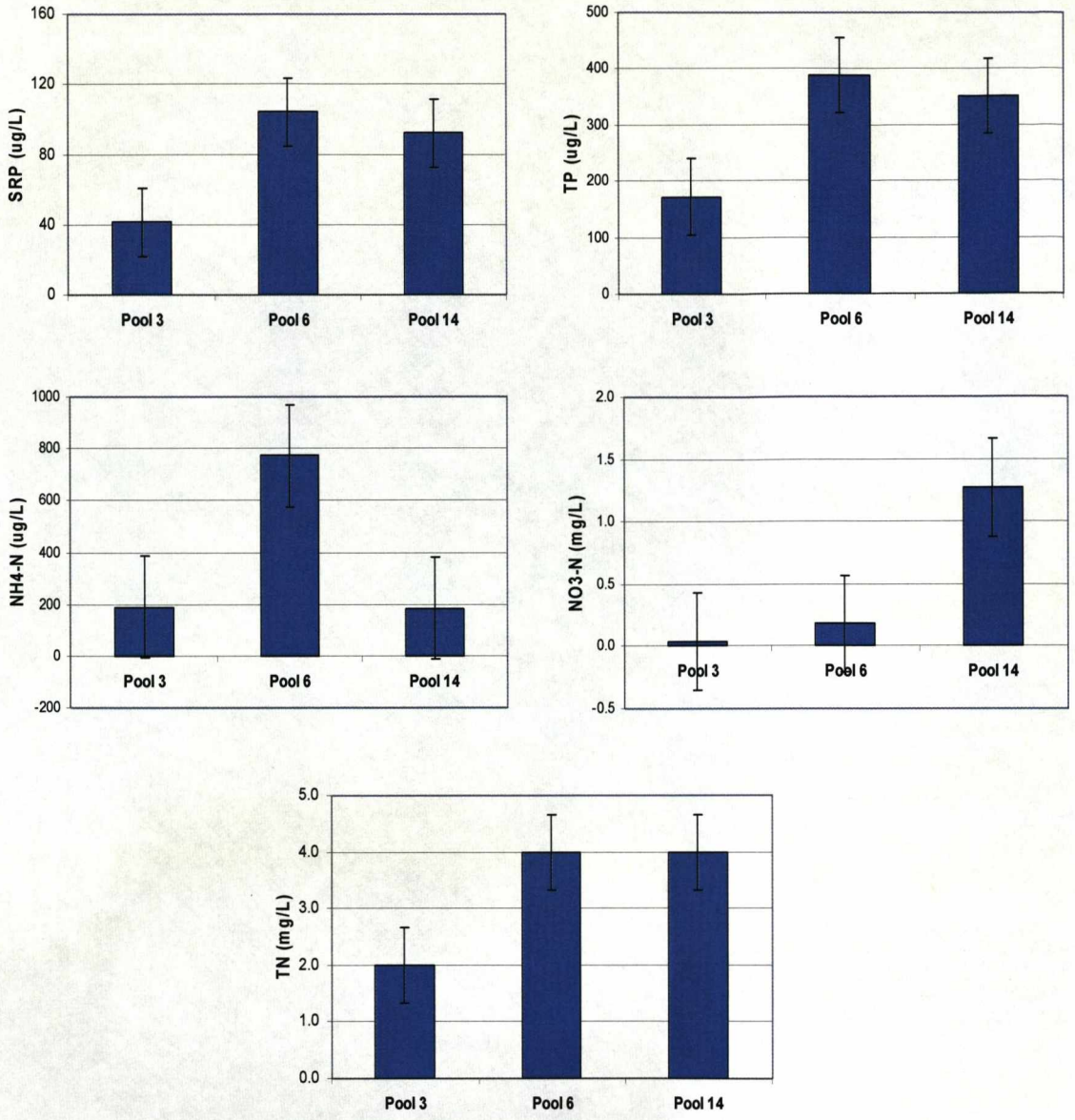


Figure 5.7 Comparison of different chemical variables in pools 3, 6 and 14 (n = 49)

In addition, TN : TP ratios in pools 3, 6 and 14 were 14, 11 and 12, respectively, meaning that nitrogen or phosphorus could be limiting in those pools (TN/TP ratio is between 10 and 17). However, in pool 14 the lake was potentially phosphorus-limited in winter (TN/TP ration is greater than 17) and in summer the lake is nitrogen-limited (TN/TP ratio is less than 10) (Florida Lakewatch, 2000). Actual TN/TP ratios of pool 14 in winter and summer were 31 and 6, respectively.

5.4.2 Biological variables

5.4.2.1 Phytoplankton

The highest concentration of chlorophyll a was detected in pool 6 and the lowest value was in pool 3 (*Figure 5.8*). Average concentrations of chlorophyll a, followed by SD (n = 24) measured in pools 3, 6 and 14 were 44±46, 100±97 and 93±48 μg L^{-1}, respectively. Results of chlorophyll a values from the studied 3 pools also confirmed that the studied lakes were hypereutrophic where concentrations of chlorophyll were higher than 40 μg L^{-1}(Florida Lakewatch, 2000)

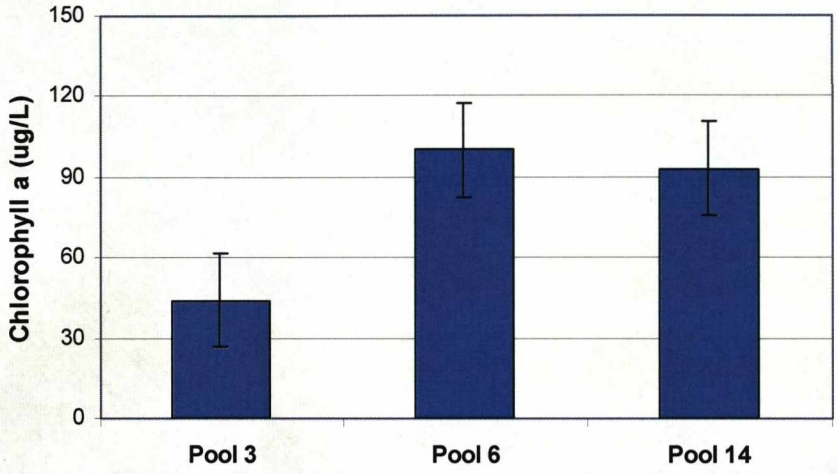


Figure 5.8 Average concentrations of chlorophyll a in pools 3, 6 and 14 with standard error (n = 49)

Statistical analysis indicated that concentrations of chlorophyll a in pool 3 significantly differed from those in pools 6 and 14 ($P_{3,6}$ = 0.017 and $P_{3,14}$ = 0.004). However, no significant difference of chlorophyll a concentrations between pools 6 and 14 was found ($P_{6,14}$ = 0.929). I also found significant differences between concentrations of chlorophyll a in different seasons in pool 3 (F = 4.659, P = 0.013) and 6 (F = 6.841, P = 0.003) but not in pool 14 (F = 0.425, P = 0.737) (*Figure 5.9*).

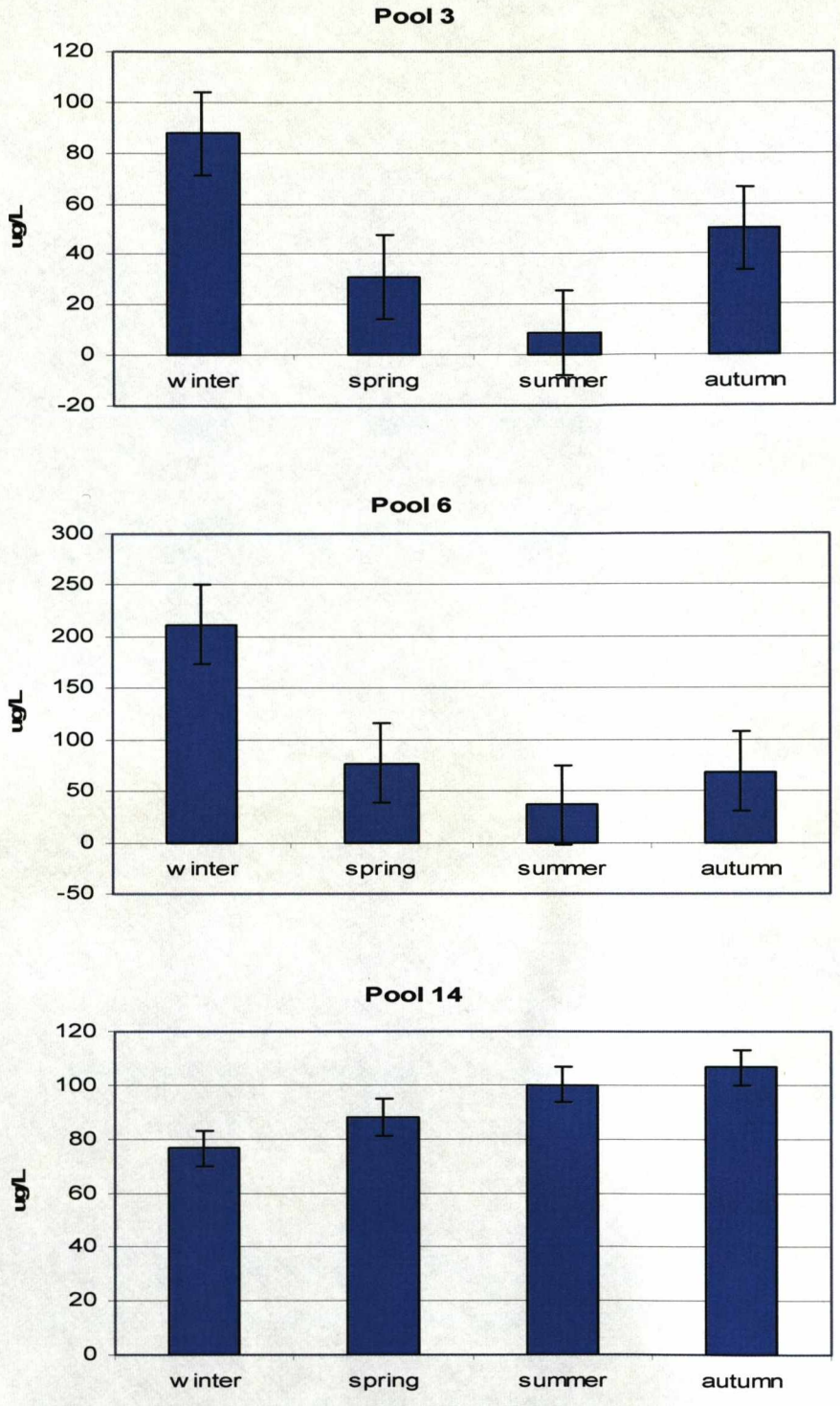


Figure 5.9 Seasonal concentrations of chlorophyll a in pools 3, 6 and 14 with standard error (n = 12)

Figure 5.10 illustrates seasonal dynamics of concentrations of chlorophyll a in pools 3, 6 and 14. Chlorophyll a concentrations in all studied pools tended to increase and remained high in winter 2005 and 2006 and then declined and were lower in summer. The highest concentration of chlorophyll a was detected in pool 6 in February 2006 with a value of 337 ($\mu g\ L^{-1}$) and the lowest value of chlorophyll a was 1 $\mu g\ L^{-1}$ in pool 3 in May 2005.

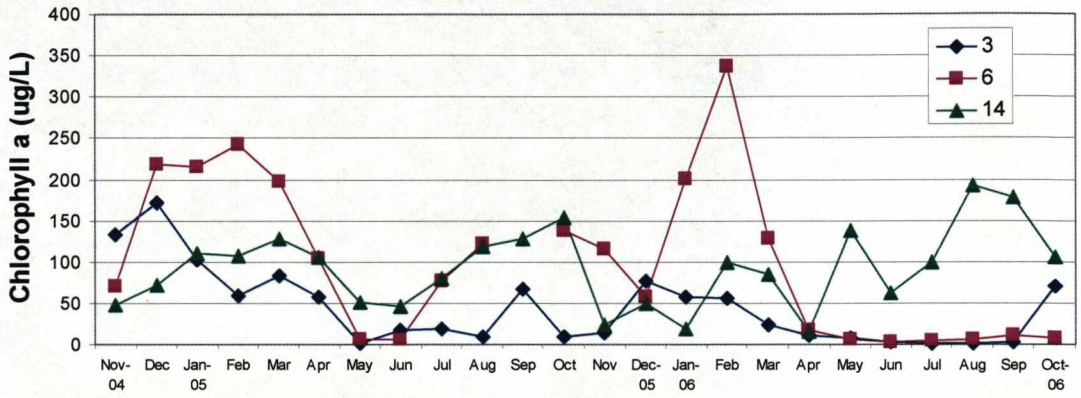


Figure 5.10 Seasonal variations of chlorophyll a concentrations in pools 3, 6 and 14

Correlation coefficients showed that seasonal variation of chlorophyll a concentrations in pool 3 significantly correlated with those in pool 6 ($r = 0.520$, $P = 0.011$). However, no significant correlations of concentrations of chlorophyll a between pools 3 and 14 ($r = -0.165$, $P = 0.441$), and between pools 6 and 14 ($r = -0.007$, $P = 0.974$) were found.

I also calculated correlation coefficients between concentrations of chlorophyll a and nutrients in the studied pools and the results showed that in pool 3, concentrations of chlorophyll a did not correlate with any nutrients (*Table 5.3*). But in pool 6 chlorophyll a concentrations positively and significantly correlated with nitrate ($r = 0.424$, $P = 0.044$) and negatively and significantly correlated with SRP ($r = -0.448$, $P = 0.042$). In pool 14, negative correlation between concentrations of chlorophyll a and NO_3^--N ($r = -0.471$, $P = 0.020$) was found.

Table 5.3 Correlation coefficients between chlorophyll a and nutrients (n = 49) in

Pool 3

Variables	Chl-a	SRP	TP	NH_4^+-N	NO_3^--N
SRP	-0.179				
TP	0.219	-			
NH_4^+-N	-0.234	0.730**	0.618**		
NO_3^--N	0.347	-0.252	-0.177	-0.173	
TN	0.238	0.130	0.347	-	-

Pool 6

Variables	Chl-a	SRP	TP	NH_4^+-N	NO_3^--N
SRP	-0.448*				
TP	0.094	-			
NH_4^+-N	-0.295	0.746**	0.724**		
NO_3^--N	0.424*	-0.227	0.020	-0.189	
TN	0.358	0.130	0.393	-	-

Pool 14

Variables	Chl-a	SRP	TP	NH_4^+-N	NO_3^--N
SRP	0.184				
TP	0.375	-			
NH_4^+-N	0.285	0.312	0.036		
NO_3^--N	-0.471*	-0.490*	-0.779**	0.029	
TN	-0.359	-0.286	-0.638**	-	-

Remark; **. = Correlation is significant at the 0.01 level and

*. = Correlation is significant at the 0.05 level and - = Auto correlated.

5.4.2.2 Zooplankton

Figure 5.11 shows composition of zooplankton in pools 3, 6 and 14. There were 4 main groups of zooplankton (rotifers, cladocerans, copepods and ostracods) found in pools 3, 6 and 14 (*Figure 5.11*). Zooplankton in pool 3 comprised rotifers 46%, copepods 30%, large cladocerans (> 1mm such as *Daphnia*) 20%, small cladocerans (< 1 mm such as *Bosmina, Chydorus, Macrothrix*) 3% and ostracods 1%. In pool 6, the main groups of zooplankton were copepods (55%), rotifers (31%) large cladocerans (11%), small cladocerans (2%) and ostracods (1%), and in pool 14, the rotifers were most abundant (52.6%) and followed by copepods (24%), small cladocerans (21.6%), large cladocerans (1.5%), and ostracods (0.2%).

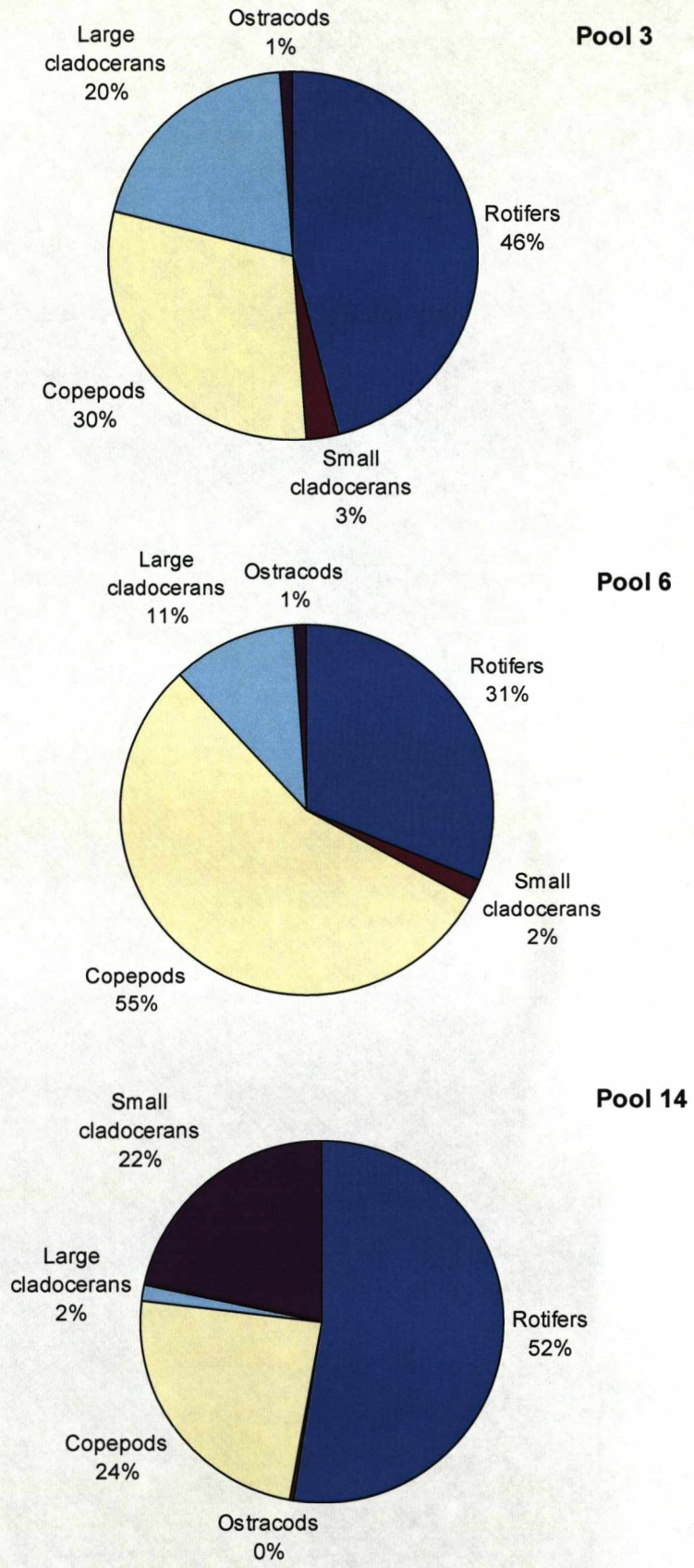


Figure 5.11 Composition of zooplankton in pools 3, 6 and 14

The rotifers, copepods and small cladocerans were most abundant in pool 14 whereas the ostracods and large cladocerans were found in the greatest numbers in pool 3 (*Figure 5.12*). Numbers of rotifers, copepods and small cladocerans were similar in pools 3 and 6. Furthermore, the least numbers of rotifers and small cladocerans were found in pool 6 whereas large cladocerans were present in the lowest numbers in pool 14.

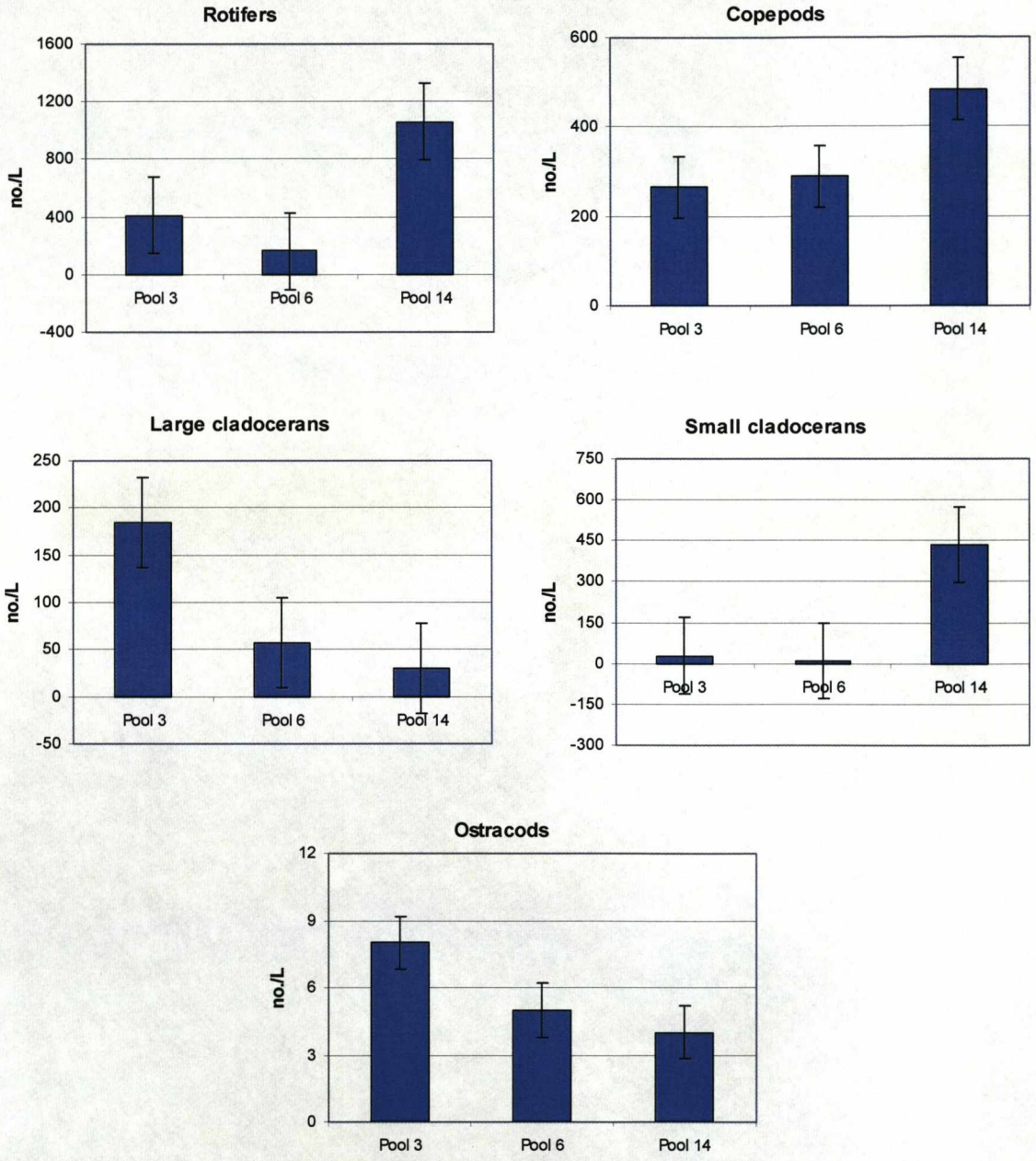


Figure 5.12 Comparative quantities of zooplankton in pools 3, 6 and 14 with standard error (n = 13)

Numbers of rotifers ($P_{3,6}$ = 0.818, $P_{3,14}$ = 0.264 and $P_{6,14}$ = 0.094), copepods ($P_{3,6}$ = 0.978, $P_{3,14}$ = 0.211 and $P_{6,14}$ = 0.287) and ostracods ($P_{3,6}$ = 0.681, $P_{3,14}$ = 0.527 and $P_{6,14}$ = 0.854) in pools 3, 6 and 14 were not significantly different. However, overall statistical analysis indicated significant differences of numbers of large cladocerans (F = 5.162, P = 0.017) and small cladocerans (F = 3.922, P = 0.039) between the pools. Numbers of large cladocerans in pool 3 were significantly different from pool 14 ($P_{3,14}$ = 0.020).

Figure 5.13 illustrates seasonal dynamics of zooplankton in pools 3, 6 and 14. Copepods in all pools started to increase in late spring and then gradually declined in autumn. Numbers of rotifers in all pools also increased in summer around June and July 2006. Large cladocerans in pools 6 and 14 appeared to increase in late spring but in pool 3, numbers of large cladocerans increased about one month later than in pools 6 and 14. Small cladocerans in pools 3 and 6 tended to increase in summer and remained high until October 2006. But in pool 14, small cladocerans were most abundant during late spring to early summer and then the population started to decline. Ostracods in pool 6 increased from May to July 2006 whereas ostracods in pool 14 were present in high numbers later in late summer around August and then decreased from autumn onward. Overall, zooplankton in pools 3 and 6 slightly increased from spring onward but in pool 14, zooplankton increased dramatically from April until September 2006. Populations of zooplankton in pools 6 and 14 tended to decrease in September toward the winter months. On the contrary, zooplankton in pool 3 increased in September but was low in winter.

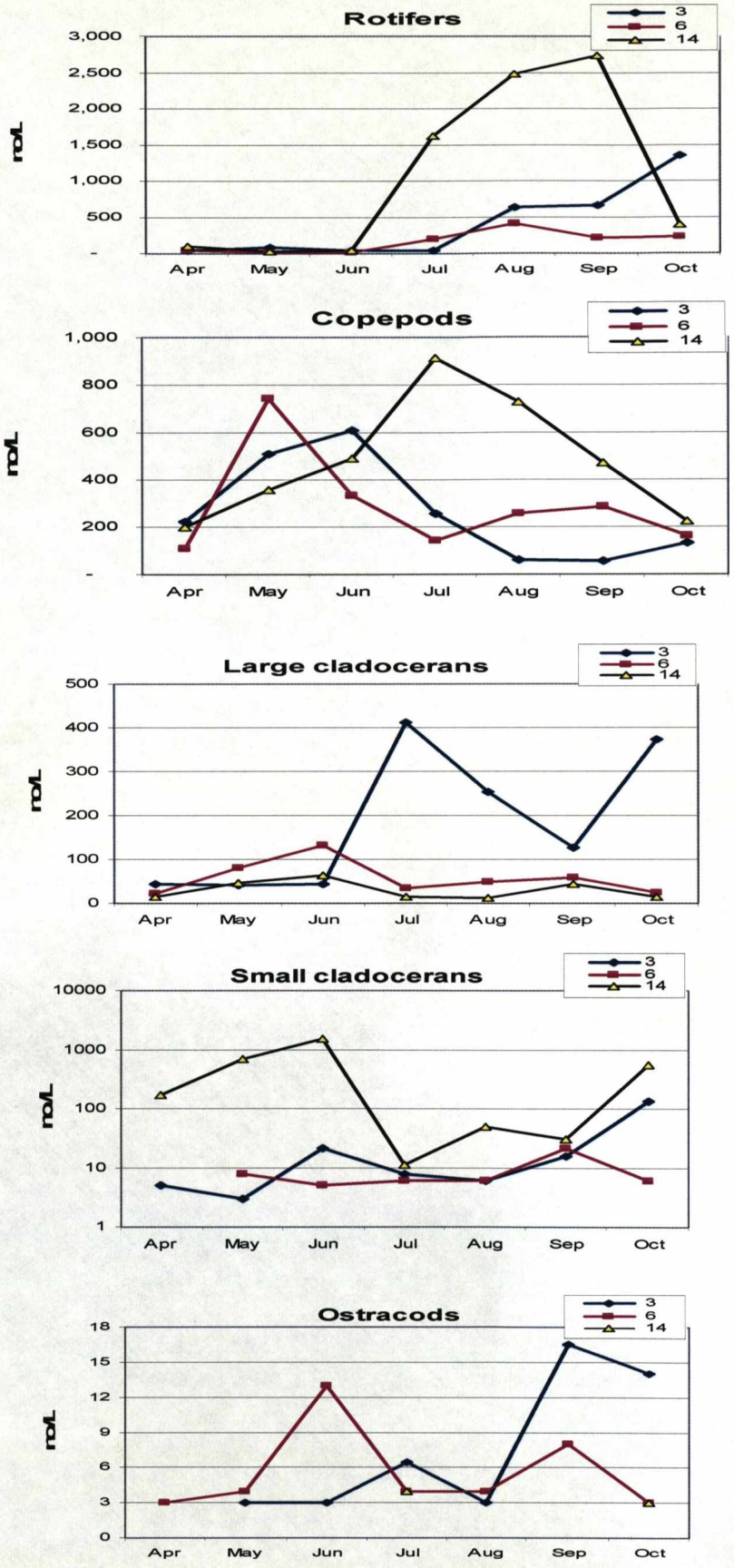


Figure 5.13 Seasonal dynamics of zooplankton (no. L^{-1}) in pools 3, 6 and 14 in 2006

Major groups of cladocerans in pools 3 were similar to those in pools 6 (*Figure 5.14*). In contrast, species composition of cladocerans in pool 14 was different from those in pools 3 and 6. In pools 3 and 6, the dominant species of cladocerans was *Daphnia* whereas in pool 14 *Bosmina* was most abundant. Smaller cladocerans (*Chydorus, Bosmina and Macrothrix*) were less abundant in pools 3 and 6 whilst large cladocerans (*Daphnia*) were not dominant in pool 14.

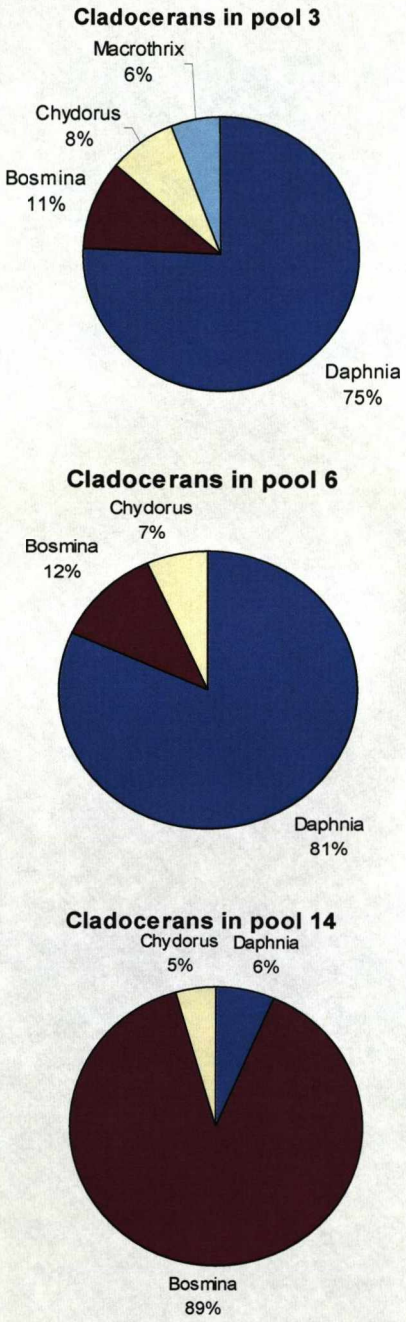


Figure 5.14 Species composition of cladocerans in pools 3, 6 and 14

Bray-Curtis analysis showed that cladoceran species in pool 3 were similar to those in pool 6 rather than pool 14 (*Figure 5.15*). Similarity of cladocerans between pools 3 and 6 was 47.5% and between pools 3 and 14 was 18.62 %. Similarity of cladocerans between pools 6 and 14 was 16.23%.

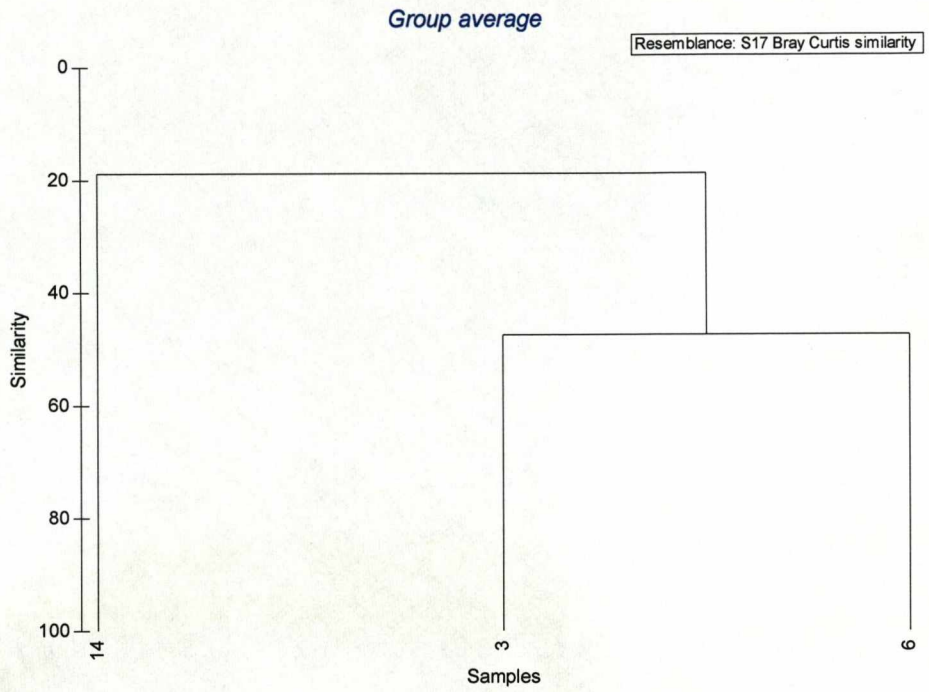


Figure 5.15 Bray-Curtis analysis of cladocerans among the pools

5.4.2.3 Macro invertebrates

5.4.2.3.1 Macro invertebrates associated with plants

Major groups of macro invertebrates associated with plants in pools 3, 6 and 14 included cladocerans (*Daphnia, Simocephalus, Bosmina, Chydorus*), pond shrimps (*Gammarus* sp*)*, water boatmen (Corixidae), non biting midges (*Chironomus, Tanypus*), mayflies (Baetidae). Both univalve and bivalve molluscs (*Planorbis, Limnaea, Sphaerium*) were present in pools 3 and 6. Other species of macro invertebrates associated with plants were caseless and cased caddis fly larvae, stonefly nymphs (Plecoptera), water beetles (Dytiscidae), leeches (*Helobdella stagnalis* (Linnaeus), *Theromyzon tessulatum* (O.F. Müller), *Erpobdella octoculata* (Linnaeus), pond skaters (*Gerris*), damselflies and dragonfly nymphs (Anisoptera and Zygoptera) and larvae of *Culex*, which were less abundant.

Numbers of plant-associated macroinvertebrates in pools 3, 6 and 14 were different. Mayflies such as *Cloeon* were most abundant in pool 3 and less abundant in pools 6 and 14 whereas *Gammarus, Chaoborus,* and molluscs were mostly found in pools 3 and 6 (*Figure 5.16*). Water boatmen were present in high numbers in pools 6 and 14 whereas non-biting midges were dominant in pools 3 and 14 and less abundant in pool 6.

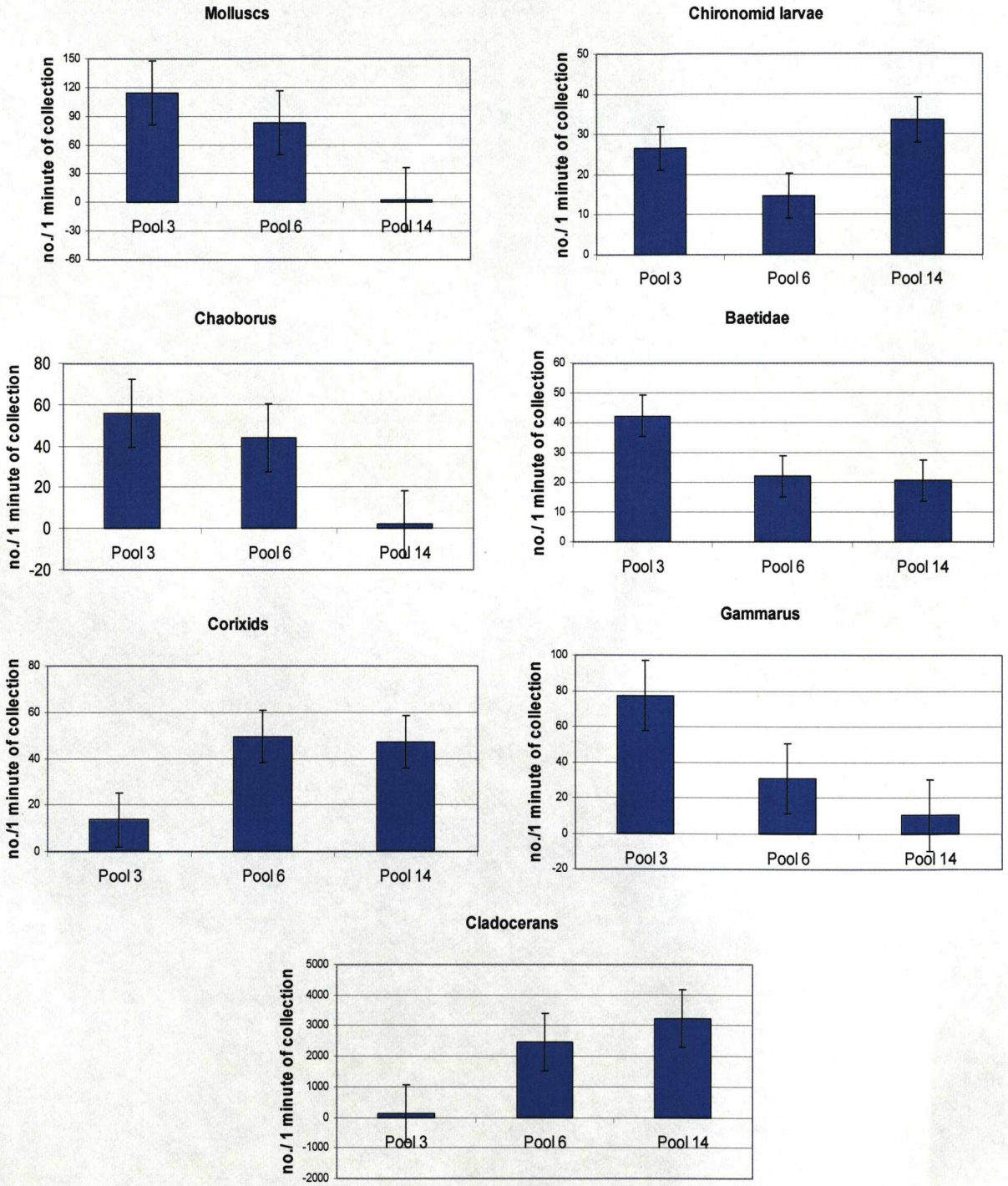
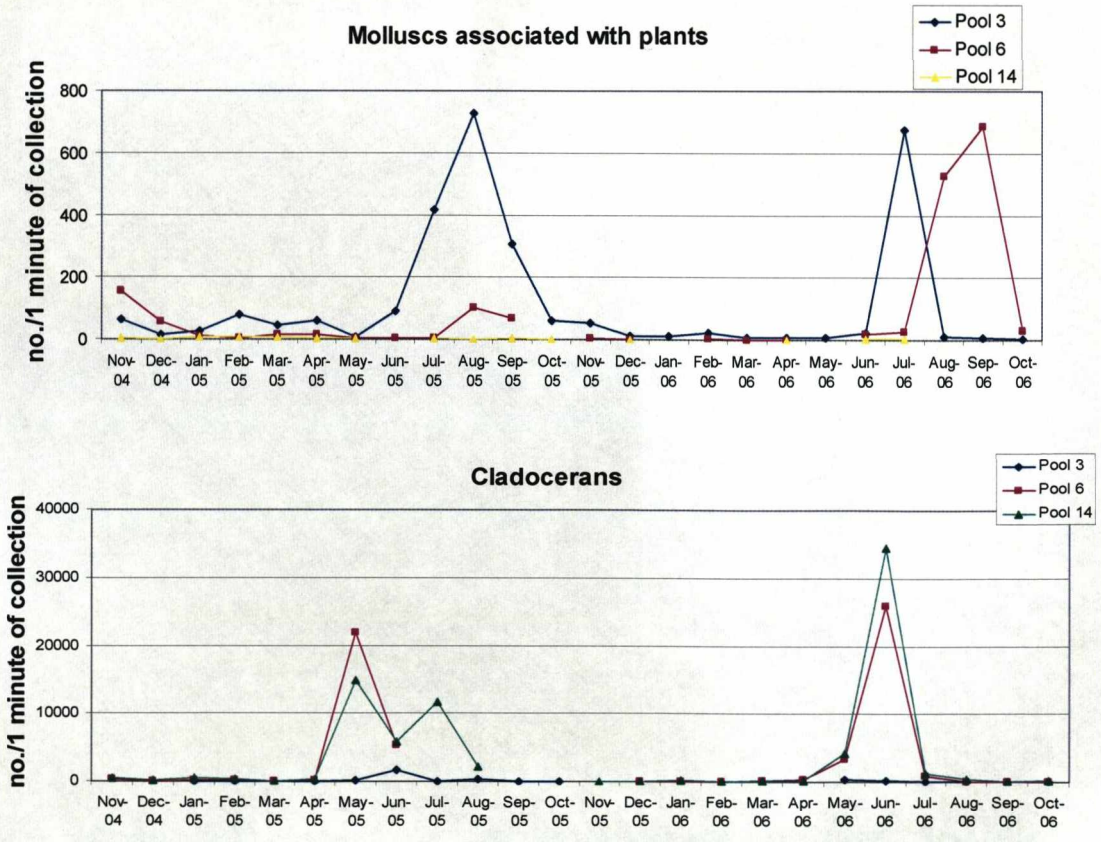


Figure 5.16 Monthly average numbers of macroinvertebrates associated with plants in pools 3, 6 and 14 with standard error (n = 5)

Tukey statistical analysis revealed that there were no significant differences between numbers of gastropods ($P_{3,6}$ = 0.809, $P_{3,14}$ = 0.120 and $P_{6,14}$ = 0.346), *Chaoborus* ($P_{3,6}$ = 0.954, $P_{3,14}$ = 0.663 and $P_{6,14}$ = 0.802), non-biting midges ($P_{3,6}$ = 0.674, $P_{3,14}$ = 0.854 and $P_{6,14}$ = 0.368), Baetidae ($P_{3,6}$ = 0.235, $P_{3,14}$ = 0.056 and $P_{6,14}$ = 0.994), water boatmen and cladocerans ($P_{3,6}$ = 0.449, $P_{3,14}$ = 0.445 and $P_{6,14}$ = 0.995) in pools 3, 6 and 14. Also, no significant differences of numbers of *Gammarus* between pools 3 and 6 were found ($P_{3,6}$ = 0.066). However, statistical analysis indicated significant differences of numbers of *Gammarus* between pools 3 and 14 ($P_{3,14}$ = 0.005). Furthermore, chlorophyll a concentrations were not correlated with numbers of cladocerans associated with plants (r_{pool3} = -0.242, P_{pool3} = 0.349 and r_{pool6} = -0.365, P_{pool6} = 0.125 and r_{pool14} = -0.188, P_{pool14} = 0.403).

Populations of macroinvertebrates associated with plants in pools 3, 6 and 14 changed seasonally. The results showed that numbers of major groups of plant associated invertebrates such as gastropods, cladocerans, midges and water boatmen associated with plants were low in winter but tended to increase in spring and summer (*Figure 5.17*). Changes in numbers of macro invertebrates were also consistent with their biomass measured.



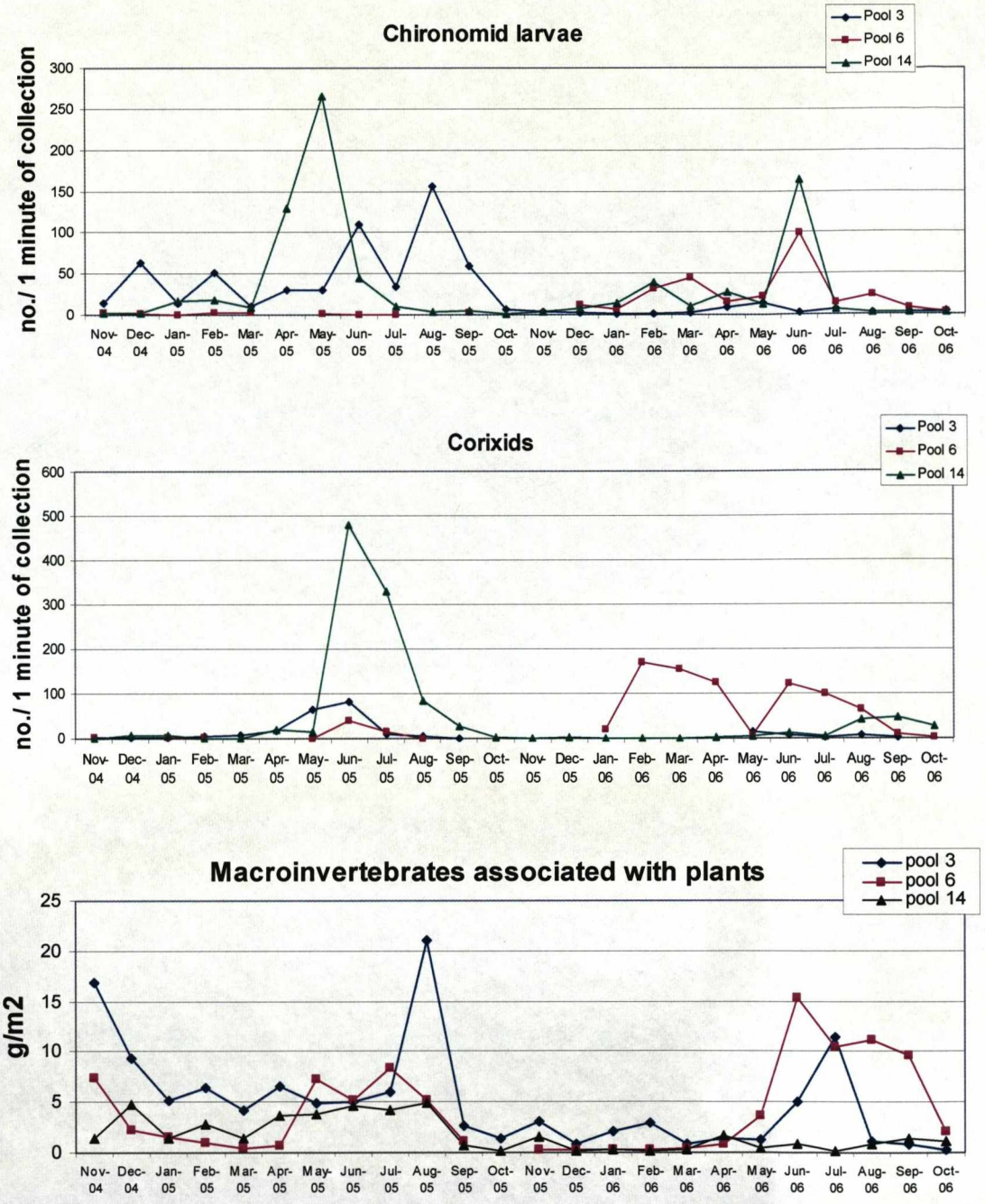


Figure 5.17 Seasonal variation of numbers of macro invertebrates associated with plants and biomass in pools 3, 6 and 14

The average biomass of macroinvertebrates associated with plants was 0.11, 0.10 and 0.04 g dry weight per one minute of collection in pools 3, 6 and 14, respectively (*Figure 5.18*). Statistical analysis indicated significant differences of biomass of macro invertebrates associated with plants between pools 3 and 14 ($P_{3,14} = 0.022$). No significant differences of biomass between pools 3 and 6 ($P_{3,6} = 0.722$) and between pools 6 and 14 were found ($P_{6,14} = 0.138$).

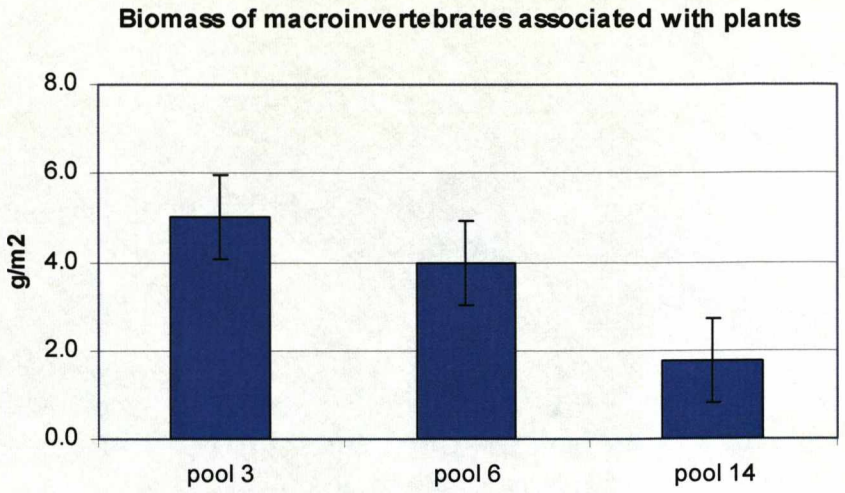


Figure 5.18 Comparative biomass of macroinvertebrates associated with plants in pools 3, 6 and 14 with standard error (n = 5)

4.2.3.2 Sedimentary macro invertebrates

Common species of benthic macroinvertebrates were biting midges (*Forcipomyia)*, non biting midges (*Chironomus, Tanypus*), oligochaete worms (*Tubiflex, Nais, Rhynchemis, Lumbriculus variegates* (Mueller) and molluscs (*Planorbis, Physa, Lymnea, Sphaerium*). Caseless and cased caddis flies, stoneflies (Plecoptera), phantom larvae (Chaoboridae), water beetles (Dytiscidae), leeches (Hirudinea), Tipulidae larvae and freshwater shrimps (*Gammarus*) were also found in some months but were low in numbers.

Forcipomyia were most abundant in pool 3 whereas oligochate worms and water hoglouse (*Asellus aquaticus* (Linnaeus)) were dominant in pool 14 (*Figure 5.19*). Numbers of molluscs were high in pools 3 and 6 compared with those in pool 14 and numbers of non biting midges were similar in all studied pools.

Statistical analysis indicated no significant differences between numbers of biting ($P_{3,6}$ = 0.152, $P_{3,14}$ = 0.084 and $P_{6,14}$ = 0.966) and non biting midges ($P_{3,6}$ = 0.878, $P_{3,14}$ = 0.870 and $P_{6,14}$ = 1.00) in pools 3, 6 and 14 (*Figure 5.18*). But I found significant differences between numbers of oligochate worms ($P_{14,3}$ = 0.001 and $P_{14,6}$

< 0.001) and molluscs ($P_{14,3}$ = 0.008 and $P_{14,6}$ = 0.001) between pools 14 and 3 and between pools 14 and 6 whereas no significant difference in oligochaete worms ($P_{3,6}$ = 0.869) and molluscs ($P_{3,6}$ = 0.739) between pools 3 and 6 was found.

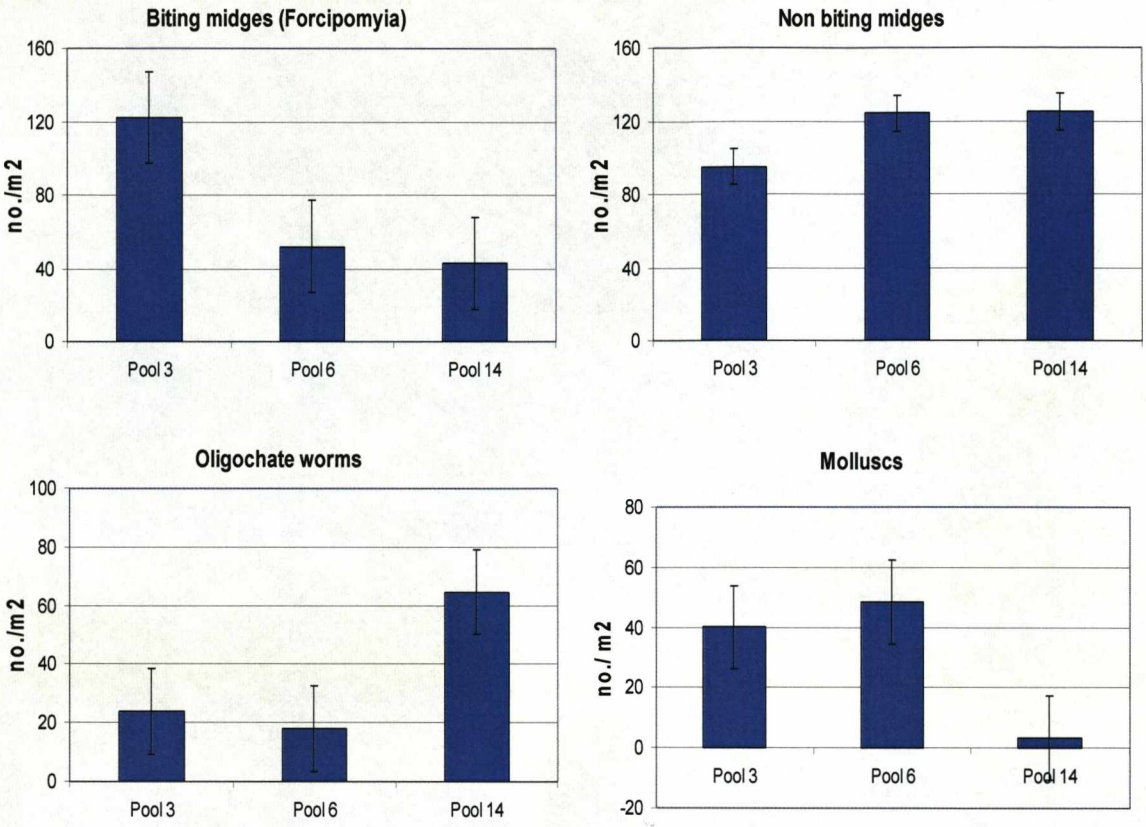


Figure 5.19 Monthly average numbers of macroinvertebrates in the sediment in pools 3, 6 and 14 with standard error (n = 5)

Figure 5.20 shows seasonal dynamics of benthic macroinvertebrates in pools 3, 6 and 14. Numbers of biting midges in pools 3 and 6 were higher in winter and spring whereas in pool 14, numbers of biting midges showed no seasonal variation. Non biting midges were most abundant in summer especially in pool 6 and then densities declined in other seasons. Densities of oligochaete worms showed large variations throughout the year but tended to increase in winter and spring. Molluscs in all pools were found in high numbers in summer and autumn and low in spring and winter.

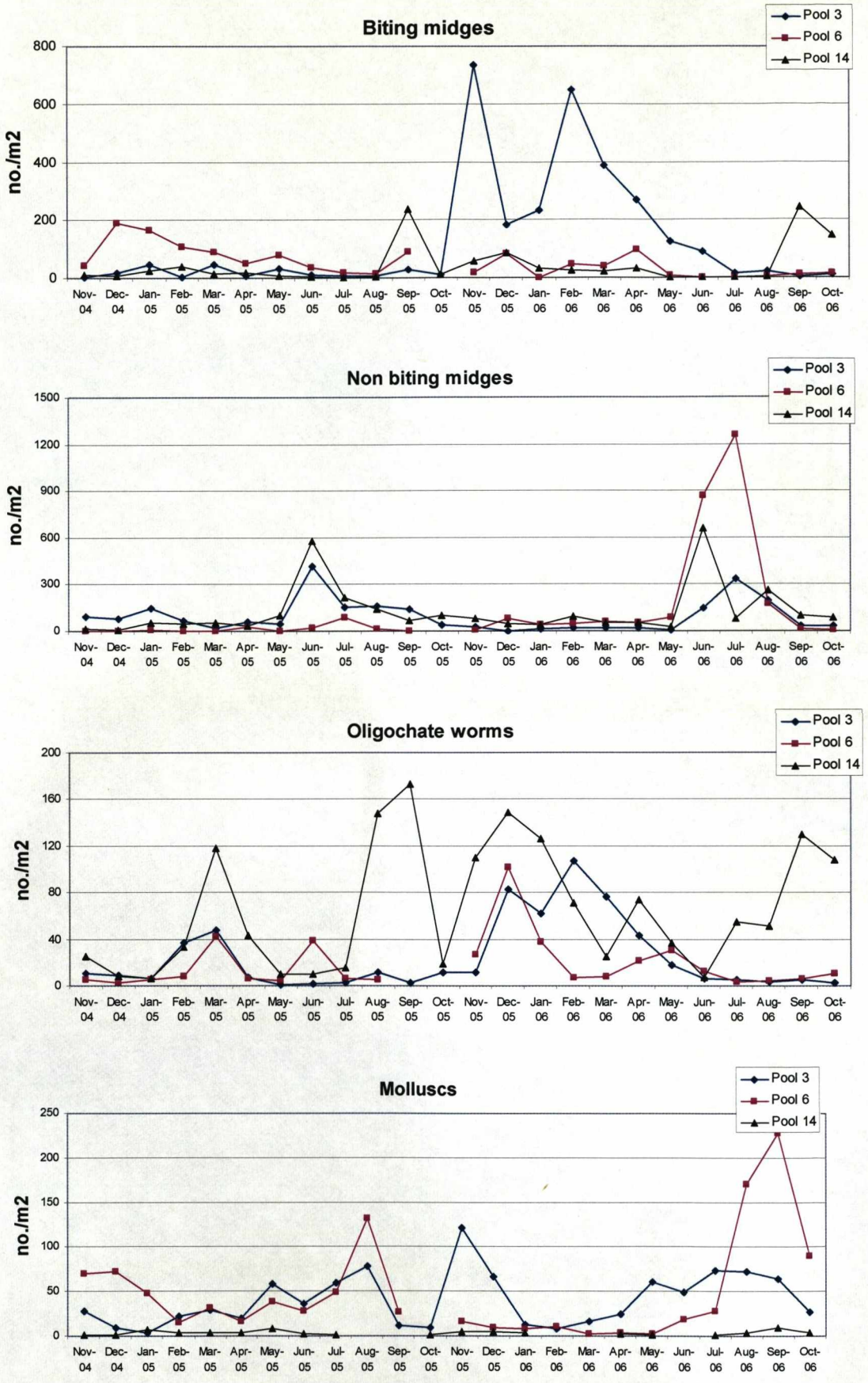


Figure 5.20 Seasonal variation of benthic macroinvertebrates in pools 3, 6 and 14

Overall, numbers of benthic macro invertebrates increased in spring and summer and decreased in winter. In summer 2005, biomass of benthic macroinvertebrates in pools 3 and 6 reached its peak. Monthly average biomass of benthic macroinvertebrates in pools 3, 6 and 14 was 2.7, 2.0 and 0.11 g dry weight m^{-2}, respectively (*Figure 5.21*).

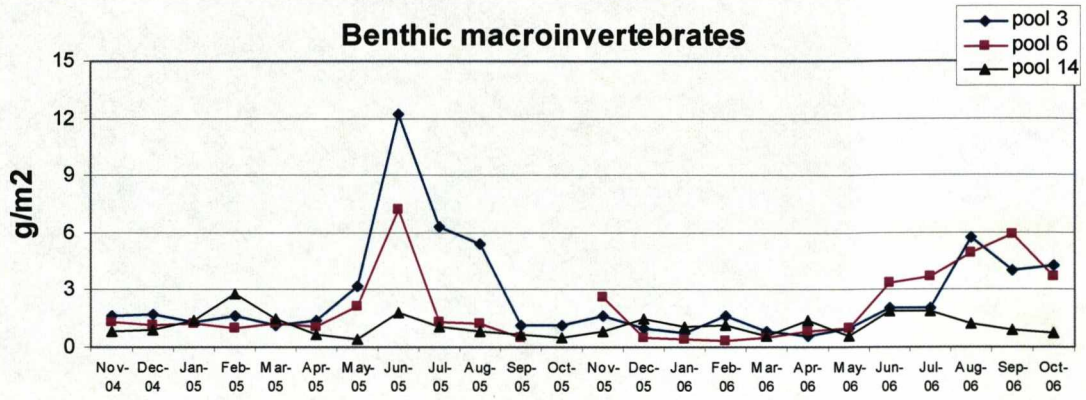


Figure 5.21 Seasonal variation of biomass of benthic macroinvertebrates

Statistical analysis indicated significant differences of biomass of macro invertebrates in the sediment between pools 3 and 14 ($P_{3,14}$ = 0.021). However, no significant differences of biomass between pools 3 and 6 ($P_{3,6}$ = 0.411) and between pools 6 and 14 were found ($P_{6,14}$ = 0.334) (*Figure 5.22*).

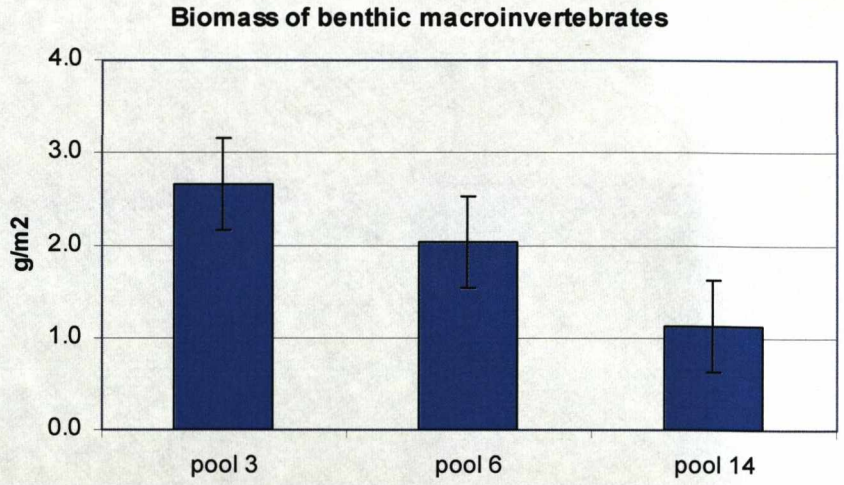


Figure 5.22 Average biomass of benthos in pools 3, 6 and 14 with standard error (n = 5)

5.4.2.4 Macrophytes

Aquatic macrophytes in pools 3, 6 and 14 were surveyed in summers 2005, 2006 and 2007 and on average, presented as mean followed by SD there were 9±3.5 (n=3), 17±0.7 (n=2) and 3 (n=3) species of macrophytes in pools 3, 6 and 14, respectively (*Figure 5.23*). Common and dominant species of aquatic vegetation in pool 3 were *Lemna trisulca* (Linneaus), *Lemna minuta* (Kunth), *Azolla filiculoides* (Lamarck), *Menyanthes trifoliate* (Linneaus), *Nymphaea alba* (Linneaus) and *Sparganium erectum. Azolla filiculoides* is an introduced species. In pool 6, common species included *Alisma plantago-aquatica* (Linneaus), *Typha latifolia*, *Polygonum amphibium*, *Ranunculus sceleratus* (Linneaus), *Alopecurus aequalis* (Linneaus). Floating plants and submerged plants (*Lemna trisulca, Lemna minuta, Spirogyra* sp., *Nitella* sp., *Drepanocladus aduncus* (Hedw.), *Potamogeton berchtoldii* (Fieber)) were also recorded in pool 6 and there was one alien species found (*Crassula helmsii*). In pool 14, *Polygonum amphibium*, *Juncus effusus* (Linneaus) and *Equisetum fluviatile* (Linneaus) were recorded. A full list of vegetation in pools 3, 6 and 14 is presented in Table 5.4.

Table 5.4 Species list of plants in pools 3, 6 and 14

Plant species	Pool 3	Pool 6	Pool 14
Floating plants			
- *Azolla filiculoides*	/		
- *Crassula helmsii*		/	
- *Lemna minor* (Linneaus)		/	
- *Lemna minuta*	/	/	
- *Lemna trisulca*	/	/	
Emergent plants			
- *Alisma plantago-aquatica*	/	/	
- *Alopecurus aequalis*	/	/	
- *Carex pseudocyperus*	/	/	
- *Equisetum fluviatile*			/
- *Juncus* sp.	/	/	
- *Juncus effusus*	/	/	/
- *Lycopus europaeus* (Linneaus)		/	
- *Menyanthes trifoliata*	/	/	
- *Nymphaea alba*	/		
- *Oenanthe aquatica* (Linneaus)	/	/	
- *Polygonum amphibium*		/	/
- *Potamogeton natan* (Linneaus)		/	
- *Ranunculus sceleratus*		/	
- *Sparganium erectum*	/		
- *Typha latifolia*	/	/	
Submerged plants			
- *Drepanocladus aduncus* (Hedw.)		/	
- Filamentous algae		/	
- *Myriophyllum* sp.		/	
- *Nitella* sp.		/	
- *Potamogeton berchtoldii*		/	
- *Potamogeton obtusifolius* (Mert. & Koch)	/		
- *Potamogeton pectinatus* (Linneaus)	/		
- *Potamogeton trichoides* (Cham. & Schltdl)	/		
- *Ranunculus circinatus* (Sibth)	/		

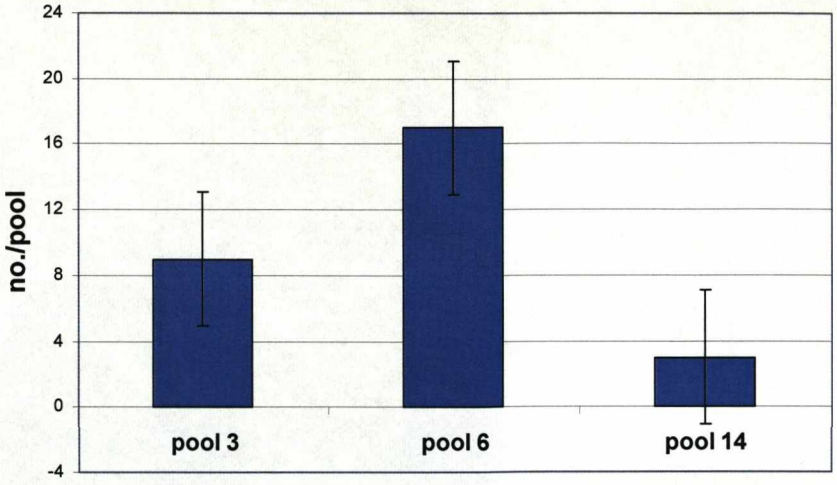


Figure 5.23 Average numbers of plant species in pools 3, 6 and 14 from 2005-2007 with standard error (n = 3)

Statistical analysis indicated significant differences in total numbers of macrophyte species between pools 14 and 3 ($P_{3,14}$ = 0.047) and between pools 14 and 6 ($P_{3,14}$ = 0.003). Significant differences of plant numbers between pools 3 and 6 ($P_{3,14}$ = 0.031) and between pools 3 and 14 were also found ($P_{3,14}$ = 0.047).

I compared the extent of macrophytes in pool 3 in summer 2005-7 and found that in 2005, *Lemna trisulca* was the main species that covered the surface area of the pool (*Figure 5.23*) together with other floating species, which were *Azolla filiculoides* and *Lemna minuta*. *Potamogeton trichoides* was the main submerged species and *Sparganium erectum* was dominant around the edge of water body. *Nymphaea alba* was also present in the south of the pool.

In 2006, there were more species of macrophytes in pool 3 than in the previous year. Water level in the pool had decreased noticeably, thus allowing amphibious plants and grasses to grow (*Figure 5.24*). In 2006, *Lemna trisulca* was still the main species but less abundant than in the previous year. Other dominant species in this year in summer included *Potamogeton pectinatus, Ranunculus circinatus, Alopecurus aequalis* and *Oenanthe aquatica*. *Sparganium erectum* was still found abundantly around the fringe of the pool and *Nymphaea alba* was still present in the south of the pool.

In 2007, the pool had received much rain and as a result water level increased and remained high in summer. *Lemna trisulca* and *Lemna minuta* were both dominant and they spread across the pool and in this year submerged macrophytes such as *Potamogeton pectinatus* and *Ranunculus circinatus* had disappeared (*Figure 5.24*). *Sparganium erectum* was still present around the edge of the pool as well as *Nymphaea alba.*

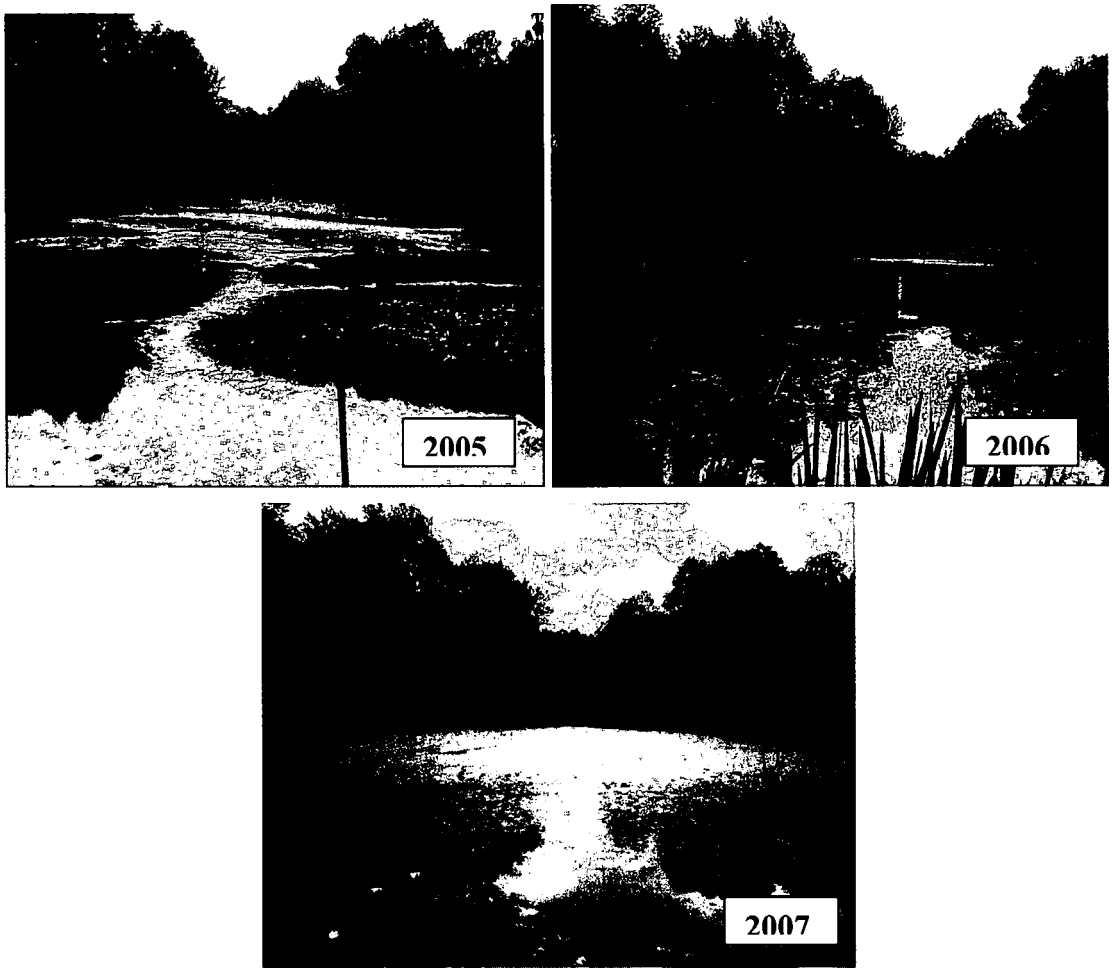


Figure 5.24 Aquatic macrophyte communities in pool 3 in summer 2005-7. These pictures were taken near the parking area, which is north-east of the pool.

I quantitatively measured growth of macrophytes using PVI values in pool 3 in 2005 - 7 (*Figure 5.25*). Overall, floating plants had the highest percentage coverage. PVI values of submerged macrophytes communities changed year by year and PVI values of some species increased while others declined. This may be the result of variation of water levels.

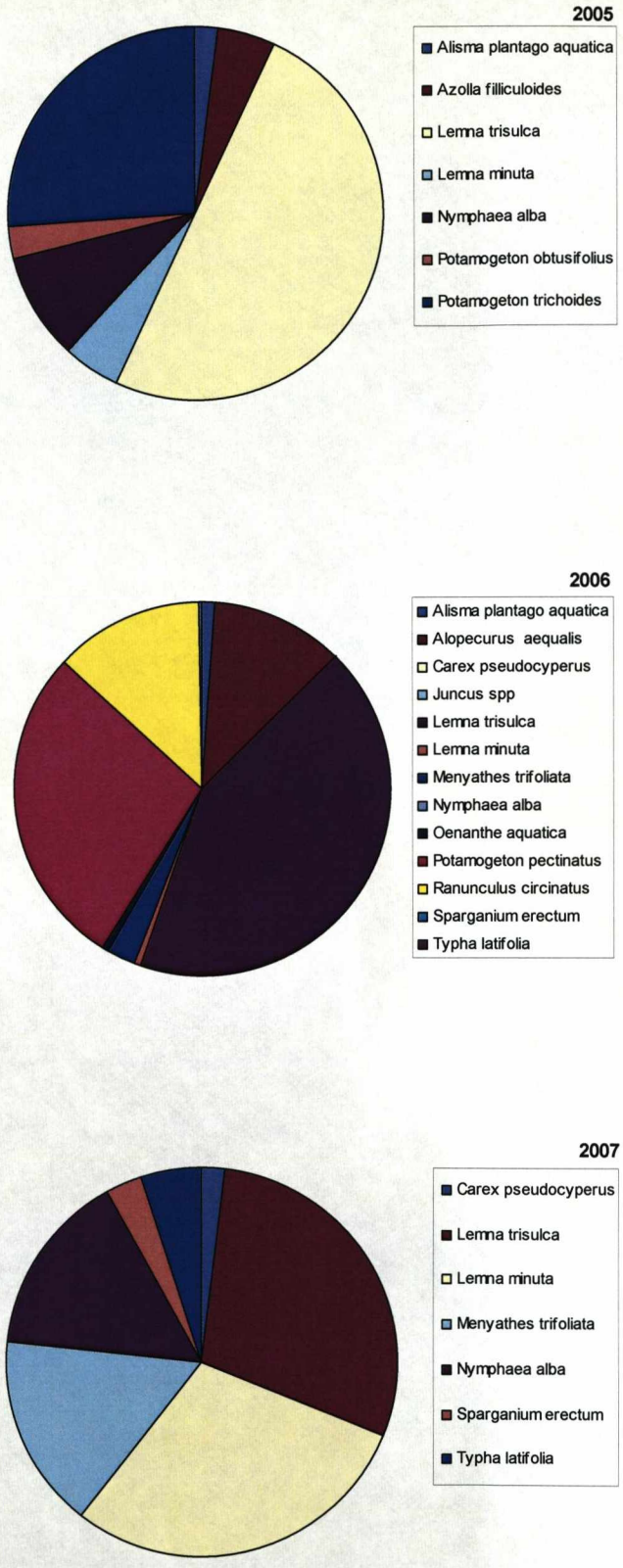


Figure 5.25 Comparative PVI values of the plants in pool 3 in 2005 -7

In pool 6 in summer 2005, water level decreased considerably and the pool had nearly dried out when a survey was done. I found only filamentous algae and *Lemna minor* and they were not abundant.

In 2006, there were more species of macrophytes present in the lake than in the previous year when the lake dried out and grasses and rushes emerged in the pool. *Nitella* sp. was most abundant and occupied about 21% of the survey area (*Figure 5.26*). *Ranunculus sceleratus, Drepanocladus aduncus* and *Alopecurus aequalis* were also found in high numbers. Marginal plants in the pool included *Typha latifolia, Juncus* sp. and *Carex pseudocyperus.* I also found floating plants such *as Lemna* spp. and *L. trisulca*, but they were not as abundant as in pool 3. And in this year, *Luronium natans*, a rare species, that has previously been absent from the site for many years was found.

In 2007, filamentous algae such as *Oedogonium, Spirogyra* and *Cladophora* were more abundant whereas *Nitella* and *Drepanocladus aduncus* declined. *Potamogeton berchtoldii* and *Polygonum amphibium* were also present in high numbers and *Typha latifolia, Juncus* sp. and *Carex pseudocyperus* were still found around the fringes of the pool (*Figure 5.27*). Other species such as *Ranunculus sceleratus, Alopecurus aequalis, Alisma plantago-aquatica, Lycopus europaeus and Oenanthe aquatica* were also found.

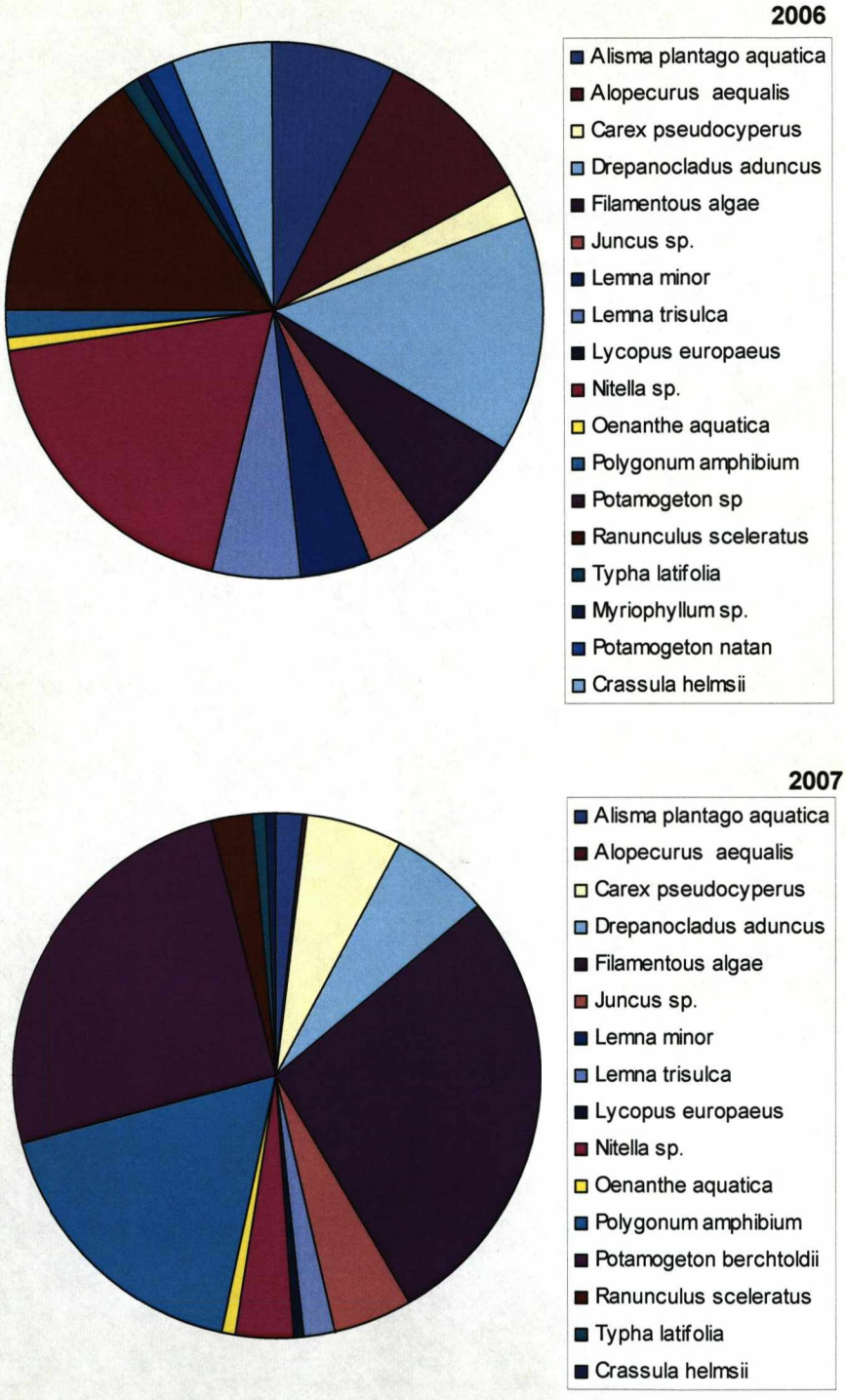


Figure 5.26 Comparative PVI values of the plants in pool 6 in 2006 -7.

Figure 5.27 Aquatic macrophyte communities in pool 6 in summer 2005-7. The pool dried out in late August 2005. (from top to bottom). These pictures were taken near main parking area (north of the pool).

In pool 14, there were only a few species of macrophytes compared with the other pools and there were no submerged macrophytes. *Polygonum amphibium* and *Juncus effusus* tended to increase from 2005 – 2007 (*Figure 5.28*). In contrast, populations of *Equisetum fluviatile* declined continuously and are likely to have been damaged by aquatic birds since summer 2005 (*Figure 5.29*).

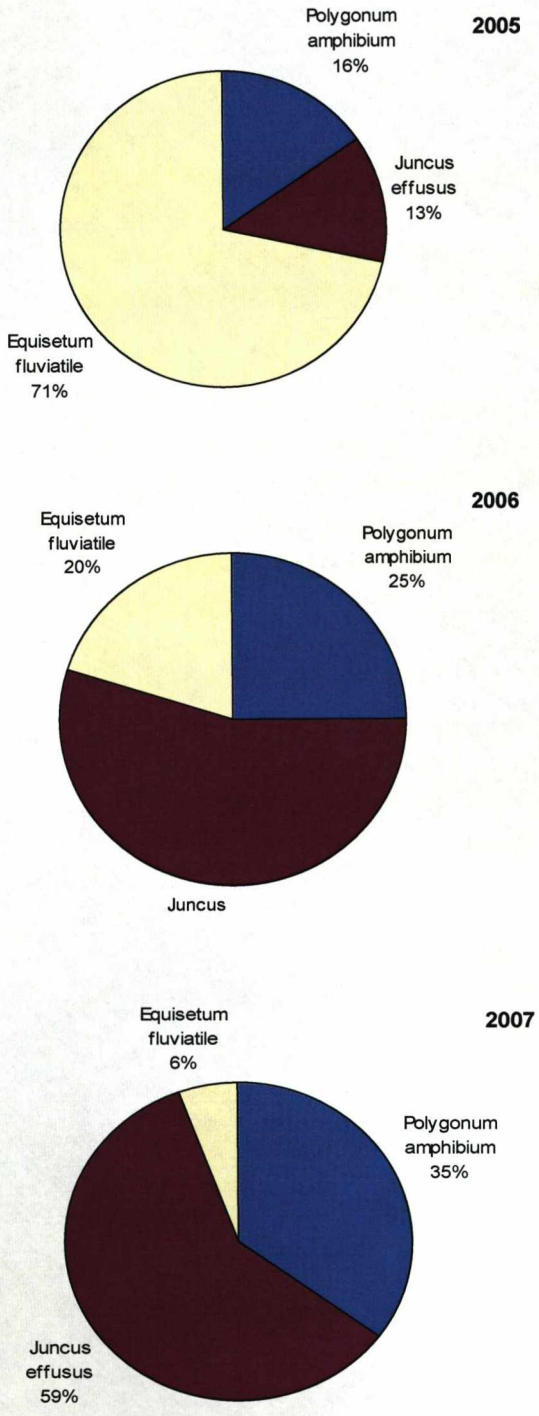


Figure 5.28 Comparative PVI values of the plants in pool 14 in 2005 -7

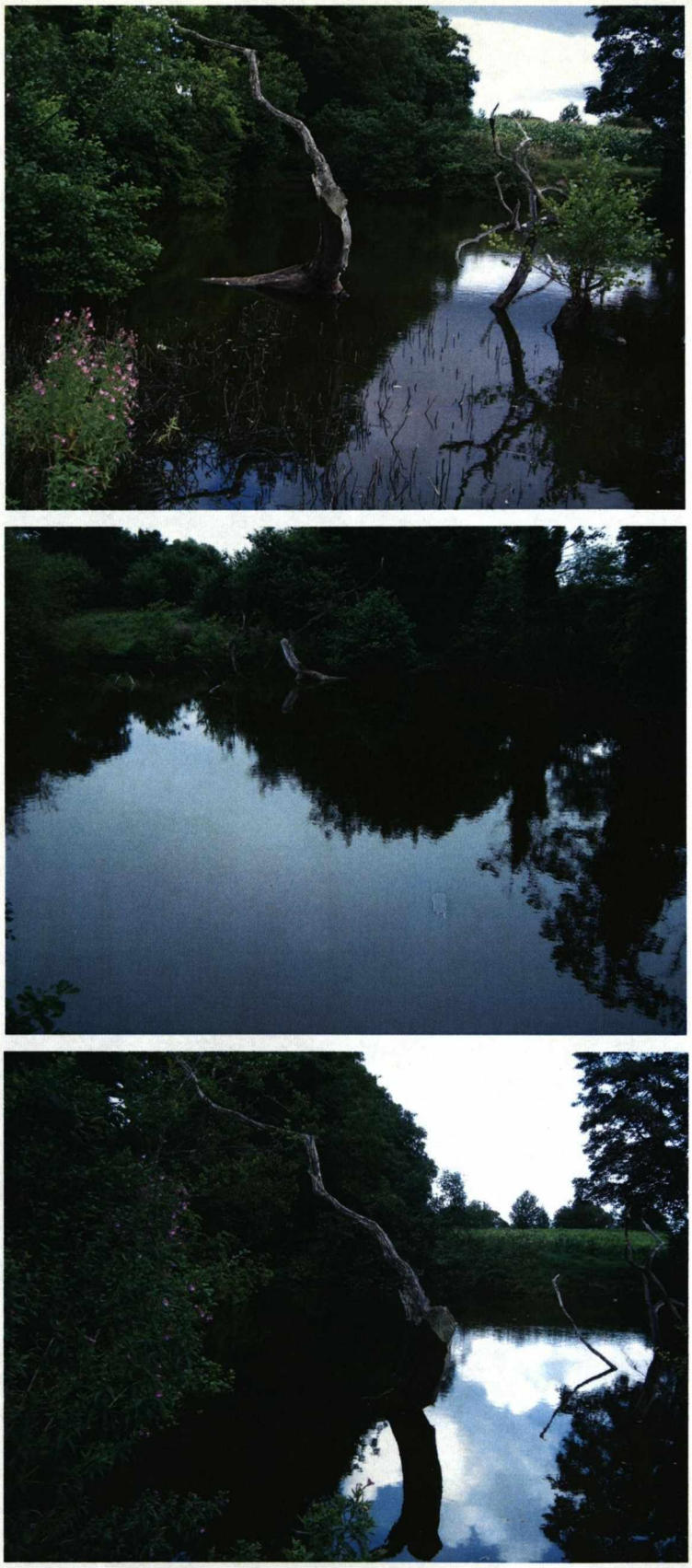


Figure 5.29 Aquatic macrophyte communities in pool 14 in summer 2005-7 (from top to bottom). Top and bottom pictures were taken from the south of the pool whereas the middle picture was taken from the north of the pool.

5.4.2.5 Aquatic birds

Greater numbers of birds were recorded in the bigger compared with the smaller pools. There were 6 species of waterfowl in pool 3, 14 species in pool 6 and 8 species in pool 14 (*Table 5.5*). The most common species of water birds recorded in all studied pools were mallard, coot, moorhen, teal and little grebe. But mallard was the only species found on the pools all year round. Bigger species of birds such as geese and swans were found in the bigger pool 6 but were not present in pool 3.

Table 5.5 Species list of birds found in pools 3, 6 and 14

Species list	**Pool 3**	**Pool 6**	**Pool 14**
Canada goose (*Branta canadensis*)		√	√
Graylag goose (*Anser anser*)		√	
Mute swan (*Cygnus olor*)		√	
Mallard (*Anas platyrhynchos*)	√	√	√
Heron (*Ardea cinerea* (Linnaeus))	√		√
Shoveler (*Anas clypeata*)		√	
Pintail (*Anas acuta*)		√	
Coot (*Fulica atra*)	√	√	√
Moorhen (*Gallinula chloropus*)	√	√	√
Tufted duck (*Aythya fuligula*)		√	
Teal (*Anas crecca*)	√	√	√
Oyster catcher (*Haematopus ostralegus* (Linnaeus))		√	
Wigeon (*Anas penelope*)		√	√
Black headed gull (*Larus ridibundus*)		√	
Little grebe (*Tachybaptus ruficollis*)	√	√	√

Average usages of waterfowl in pools 3, 6 and 14 were approximately 620, 18,800 and 3,200 bird hectare^{-1} 2 years^{-1}, respectively (*Figure 5.30*). Statistical analysis indicated significant differences of average usages of birds (bird-month hectare^{-1}) between pools 6 and 3 ($P_{6,3} < 0.001$) and between pools 6 and 14 ($P_{6,14} < 0.001$). However, no statistical difference between usages of birds between pools 3 and 14 was found ($P_{3,14} = 0.425$).

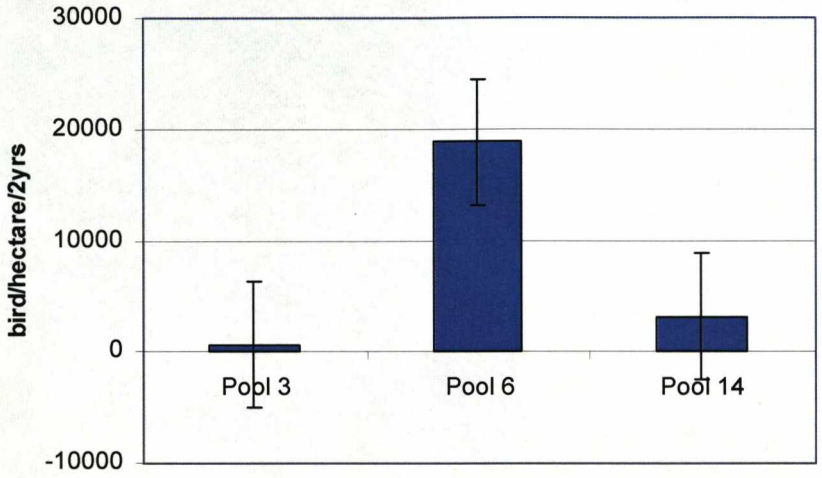


Figure 5.30 Average usages of bird in pools 3, 6 and 14 with standard error (n = 49)

In addition, average usages of birds in pool 6 were significantly different in each season (F = 7.791, P = 0.001) (*Figure 5.31*). However, no significant differences in average usages of waterfowl in each season were found for pools 3 (F = 2.936, P = 0.063) and 14 (F = 0.881, P = 0.468). Average usages of aquatic birds in pool 6 in winter (*w*) were significantly different from those in spring (*sp*) (Pw,sp = 0.001) and summer (Pw,su = 0.005).

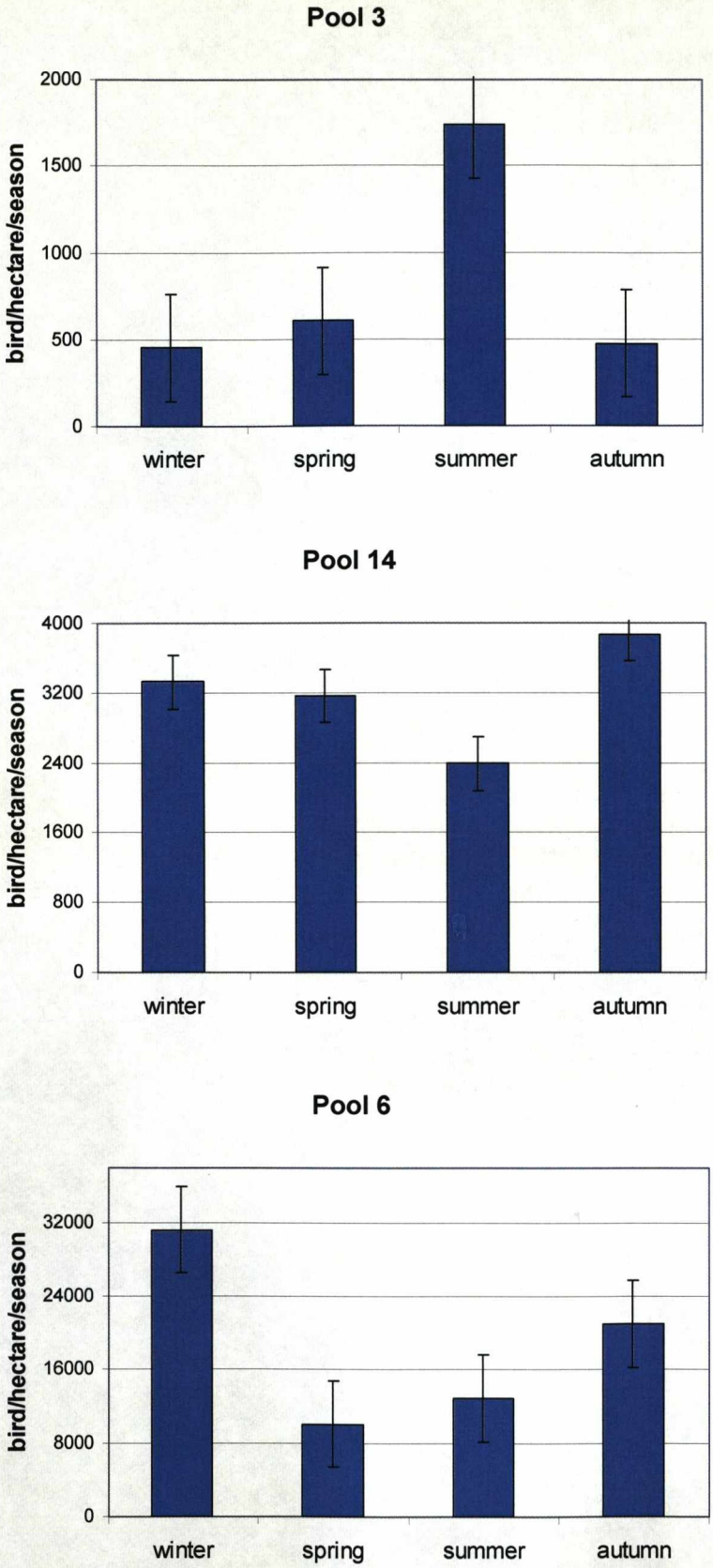


Figure 5.31 Average numbers of waterfowl in each season in pools 3, 6 and 14 with standard error (n = 12)

Densities of water birds (birds hectare^{-1}) in pools 6 and 14 increased in winter and decreased in summer (*Figure 5.32*). In pool 6, I found a large congregation of Canada geese during winter in 2004 but in 2005, teal and mallard were the most abundant species. In pool 3, densities of waterfowl were high during summer 2005 and birds were absent from the pool during winter in 2005.

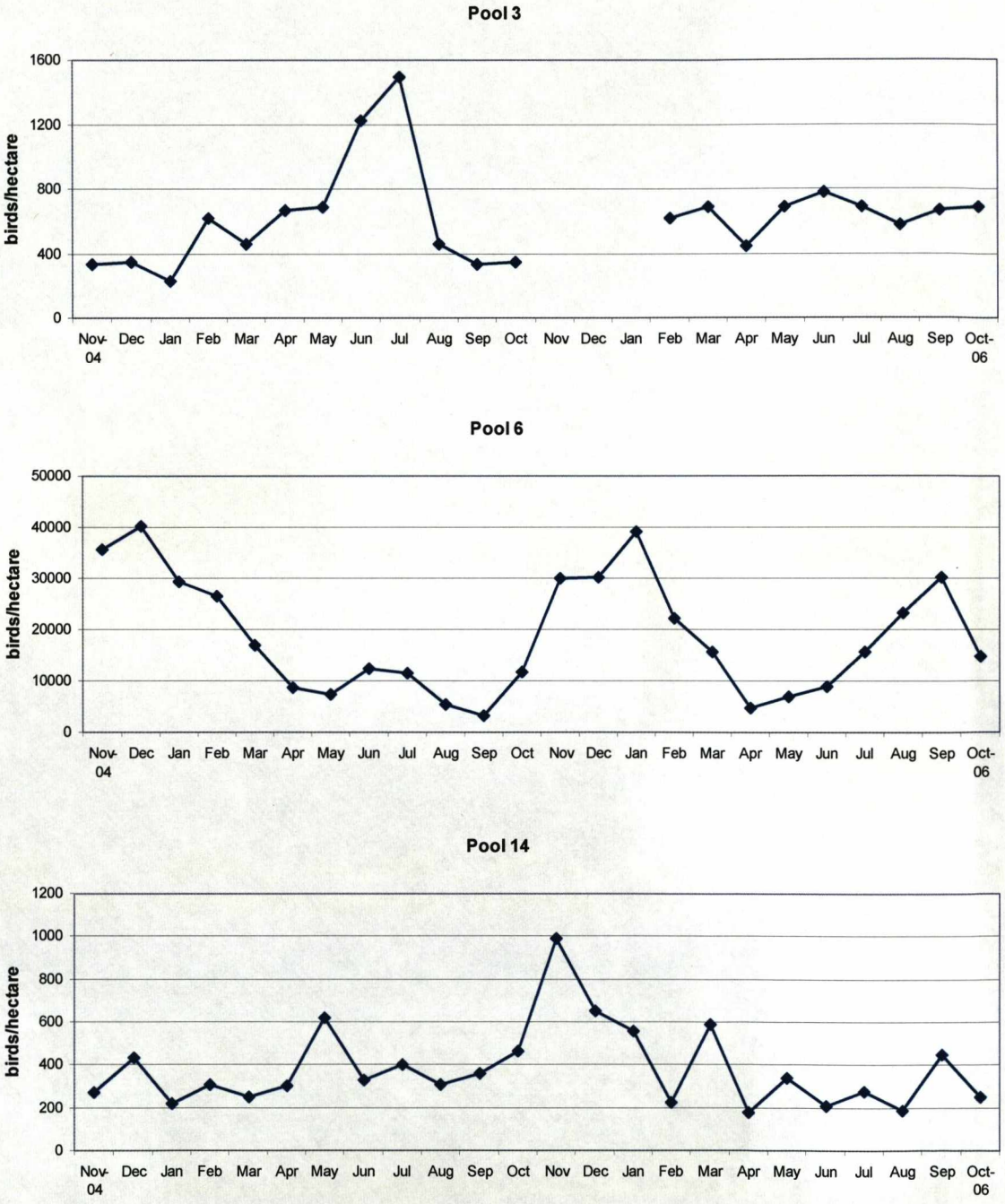


Figure 5.32 Seasonal changes of bird densities in pools 3, 6 and 14

5.5 Discussion

5.5.1 Physico-chemical properties

Location, size and depth of water bodies may result in different limnological characteristics. Physical properties of lakes such as temperature, water level and pH in pools 3, 6 and 14 changed seasonally. Pearson correlation coefficients revealed that regional factors such as the weather were likely to influence changes in water levels and temperature in the pools as those variables changed in the same pattern. Also, all pools depend on rain as the main supplier of water and that is the main reason why climate played a vital role in those pools. A study of lakes in the Windermere catchment also showed the same results that variations in the weather influenced the physical characteristic of the lakes as the lakes were all exposed to the same weather patterns (George et al. 2000). Additionally, it is noticeable that lake temperature is affected by depth of the water bodies as I found that on average, temperature of the deeper pool 14 was 2 degree Celsius lower than in shallow pools 3 and 6. Shallower lakes are warmed up faster than deeper pools as the sunlight falls aerially and the same energy flux must heat more water in a deep pool. pH values in all pools were neutral and rather constant throughout the year.

Total suspended solid (TSS), conductivity and alkalinity values were relatively high in pool 14 and low in pools 3 and 6. High total suspended solid (TSS) in pool 14 was likely due to particles brought by surface runoff and erosion from agricultural areas around the pool or high densities of phytoplankton or both. The statistical analysis showed that high TSS of pool 14 was significantly correlated with chlorophyll a concentrations (Chapter 2). As indicated by TSS values, high erosion rates originating from outside the pool may cause high conductivity in pool 14. Eilers et al. (1989) discovered that a large increase in conductivity shows an association with increases in land use development on the lake perimeter. Alkalinity of pool 14 was also high and the presence of high nitrate may be related to high alkalinity. Brewer and Goldman (1976) and Schindler (1986) stated that high rates of nitrate (NO_3^-) reduction caused by denitrification or phytoplankton uptake or both could generate an increase in alkalinity. However, this is unlikely where alkalinity values are comparatively high. In contrast, ammonium addition (NH_4^+) could decrease alkalinity markedly (Schindler et al. 1985; Schindler, 1988) and this was shown in pool 6 where ammonium

concentration was high and alkalinity was low (*Figure 5.33*). This is unlikely to be a causative relationship however as the ammonium levels were probably linked to sediment release and the low alkalinity to the acid peat soil in the vicinity.

There were distinct differences in quantities of nutrients such as nitrate, SRP and ammonium nitrogen in the pools and such differences may be influenced by regional and local factors, mainly land use in the catchment area and sediments in the pools. Local factors appeared to influence quantities of nutrients in pools 3 and 6, whereas in pool 14 a regional factor was considered crucial because the pool received nutrients from a probably bigger catchment.

Overall, as pool 3 is isolated, surrounded by mature woodland and less disturbed by aquatic birds, lower concentrations of nitrogen and phosphorus of pool 3 compared with other pools were found. Internal loading acting as a local factor seemed to be more important in this pool than external impacts as indicated by SRP and TP increasing in summer due to mineralisation from the sediment. Sondergaard et al. (2005) stated that internal processes are important and high in small lakes having no surface flows and all nutrients entering the lakes will be retained and potentially recycled with the lake.

SRP and TP concentrations in all studied pools tended to increase in summer and were lower in other seasons. SRP and TP were relatively high in pools 6 and 14 and were low in pool 3. Higher concentrations of SRP and TP in pool 6 were probably due to local factors, the birds and internal loading (Chapter 3). There were more species and numbers of birds in pool 6 and these birds may increase organic and inorganic phosphorus to the lake. The lake sediment may also be another key component causing high quantities of phosphorus in pool 6, especially in summer (Chapters 3, 4). Nitrate in pool 6 was relatively low suggesting less impact of nutrient input from the surrounding agriculture land, as indicated by the nutrient budget in Chapter 3.

Concentrations of ammonium nitrogen in pool 6 were remarkably high compared with those in pools 3 and 14. High amounts of ammonium nitrogen in pool 6 are likely due to mineralisation, low dilution capacity and aquatic birds. Experiments on nutrient release presented in Chapter 3 showed that ammonium nitrogen was released from the

sediment in summer. This result was consistent with Serruya et al. (1974) and Nowlin et al. (2005) reporting that ammonium nitrogen was released from the lake sediment due to an intense process of ammonification. Also, a dramatic decrease in water level and low dilution capacity due to high evaporation in summer may have left higher quantities of ammonium nitrogen in pool 6. Karakhanian et al. (2001) reported that in Greece progressive dramatic reduction in water in many lakes has led to eutrophication. Furthermore, in winter 2004, an increase in ammonium nitrogen concentration was correlated with high number of aquatic birds, particularly geese and several studies have supported that nitrogen contributions by aquatic birds are significant in lakes (Moss and Leah, 1982; Post et al. 1998; Olson et al. 2005).

In 2005, pool 6 completely dried out and changes in water levels both before and after wetting had an impact on water quality. High concentrations of nitrogen and phosphorus were detected after re-wetting of sediments in the pool and the reason was likely due to an initial flush of available N and P which can be incorporated into bacterial or macrophyte biomass, coupled with an increase in bacterial activities, particularly nitrification (Baldwin and Mitchell, 2000; Scholz et al. 2002). Baldwin and Mitchell (2000) stated that inundation of soils will result in the liberation of C, N and P from the sediments and consequently, an increase in anoxic bacterial processes such as P release and denitrification.

In pool 14, high concentrations of SRP, TP and nitrate could be possibly related to regional anthropogenic influence from land use. George et al. (2000) reported that long-term changes in the Windermere catchment influenced the chemical characteristics of the lakes and their general productivity. This was consistent with our results in that nitrate concentrations corresponded well with an increase of water level implying that nitrate came from surface and soil water runoff from agricultural areas around the pool. As mentioned earlier, pool 14 is surrounded by maize and grazed fields and thus surface runoff and erosion from agricultural lands may bring about N and P to the lake. Moreover, there is a grazed pasture for horses at the west end of pool 14 and wastes produced by horses may have the potential to increase N and P concentrations when transported to the lake. Sondergaard et al. (2005) stated that small lakes in the agricultural landscape have a high risk of impact from nearby farming. Furthermore, Daniel et al. (1998) and Sharpley et al. (1993) found that long-

term land application of P as fertilizer and animal wastes has resulted in elevated levels of soil P in many locations in the USA. The study of fossil pigments deposited in the bottom sediments of Lake Beskie to assess changes in the primary production by Rybak (1988) also revealed that the erosion and increase in mineral fertilization in an agricultural area led to increased leaching of nutrients into the lake which resulted in increased primary production and hypolimnetic anoxia. Impact of nutrient loading from agricultural areas on pool 14 is far more serious than in pool 6 because there is no proper riparian zone around the pool that acts as a natural bio filter to protect the aquatic environment from excessive contaminated surface runoff.

Moreover, trampling and grazing of horses near the edge of pool 14 is likely to increase surface runoff and may cause more nutrients to be transported into the pool. Pietola et al. (2005) stated that in pastures with high P status at the soil surface due to animal excreta and surface application of fertilizers, delayed filtration due to soil compaction caused by trampling can result in increased P transport in runoff. One study of the effects of grazing by horses showed that horses have a strong impact, especially in upland areas (Turner, 1987). Gleason et al. (1979) also found that heavily grazed marshes may be more susceptible to erosion and storm damage, since accretion of sediment in marshes is a function of the density of grasses present to trap particles. Therefore, improper livestock use can exert a considerable impact on stream-bank erosion and sediment transport by changing, reducing or eliminating the vegetation that borders streams (Kauffman and Krueger, 1984; Hall et al. 1999)

5.5.2 Biological variables

Shallow pools 3 and 6 are fish free whereas the deeper pool 14 contains fish. I have spoken to fishermen regularly visiting pool 14 and they told me that large carp are present. Sometimes while sampling, I also saw big fish in the pool. Because of drying up, the fish in pool 6 were removed in 1987 (Shropshire County Council, 1998) and drying up in pool 3 in some years has killed any previous fish as already mentioned in Chapter 2. Frozen water in winters and droughts in summers of shallow pools 3 and 6 have resulted formerly fish kills. In contrast, fish can survive in the deeper pool 14. A study of nearly 800 Danish lakes revealed that most shallow lakes with an area < 0.1 ha were without fish due to winter kill and droughts (Sondergaard et al. 2005). Ellis

and Stefan (1989) explained that the death of fish under ice was due to oxygen deficiency that threatens hundreds of shallow lakes in the upper Midwest of the United States every winter.

5.5.2.1 Phytoplankton

Seasonal variation of phytoplankton in the studied pools 3 and 6 showed similar trends in that phytoplankton tended to decrease in summer and increase in winter. The appearance of phytoplankton in winter could be explained by nutrient availability and the shortage of zooplankton. I found that an increase of nutrients, especially nitrate in winter corresponded well with high concentrations of chlorophyll a. Although environmental conditions in winter were not as favourable as in summer, high concentrations of nutrients may be good enough to stimulate growth of phytoplankton in pools 3 and 6. In contrast, in pool 14, biomass of phytoplankton measured as chlorophyll a was low in winter and high in summer and autumn with low zooplankton grazing potential. Changes of chlorophyll a concentrations in pool 14 were similar to those in other lakes (Foy et al. 1976; Temponeras et al. 2000). A decrease of phytoplankton in pool 14 in winter could be the result of environmental conditions such as unfavourable light and an increase in phytoplankton biomass in summer is likely partly due to increased temperature and nutrient release from the sediment that promote growth. Rhee and Gotham (1981) stated that temperature, light intensity, and day length were related to growth of phytoplankton and they found that the optimal growth temperature for *Scenedesmus* sp. ranged between 20-25°C. Foy et al. (1976) also revealed that *Oscillatoria redekei* (van Goor) reached its maximum crop in April while species such as *O. agardhii* (Gomont) were dominant during the summer months.

Normally when water bodies receive high amounts of plant nutrients, particularly those that are limiting factors, algae will grow dramatically (Florida Lakewatch, 2000; Hunt and Matveev, 2005). In pools 3 and 6, high quantities of total nitrogen and total phosphorus in summer were not reflected in chlorophyll a concentrations. Therefore, there may be other factors controlling growth of phytoplankton community in summer and one possible explanation is likely due to grazing effects of zooplankton. I suggest that a local factor, zooplankton, played a major role in determining phytoplankton

populations. Zooplankton, especially *Daphnia* was most abundant in pools 3 and 6 in summer and may control populations of phytoplankton. Sommer (1985) found that phytoplankton edibility declines after the zooplankton maximum with a time lag of about one month and in summer grazing pressure keeps the edible algae at a low biomass. Moss et al. (1997) stated that *Daphnia* is an efficient grazer and a moderate sized population of *Daphnia* dispersed in a lake can filter the entire lake volume more than once a day at equable temperatures. Quiblier-Lioberas et al. (1996) also revealed that the highest grazing impact of zooplankton on phytoplankton community in Lake Pavin, France, occurred in summer and was associated with very low phytoplankton biomasses. *Daphnia* can, therefore, effectively prevent a phytoplankton community of any significance from developing (Lampert et al. 1986).

High and low grazing effect of zooplankton on phytoplankton could also be related to temperature. Figure 9 implies that there was a lot of grazing in summer compared with that in winter in pools 3 and 6. Burns (1969) discovered that filtering rate of zooplankton such as *Daphnia magna* (Straus) increased with increasing temperature and at 20 and 25 ^{o}C was more than twice the rate at 15 ^{o}C. McMahon (1965) also reported that filtering rates of *D. magna* (Straus) increased with increasing temperatures up to 28^{o}C, above which rates declined rapidly.

Low densities of large crustacean filter feeders, primarily *Daphnia,* as an effect of fish predation may result in low pressure of grazing on phytoplankton in pool 14. Smaller zooplankton such as copepods and rotifers were abundant in pool 14 and these species are less efficient in the control of phytoplankton. Quiblier-Lioberas et al. (1996) revealed that the grazing impact of the copepod *Acanthodiaptomus* sp. was relatively low (< 17% day^{-1}) compared with that of cladocerans with the greatest grazing impacts up to 36% day $^{-1}$. Mourelatos (1988) and Lampart et al. (1986) found higher values for grazing impacts of the entire zooplankton community on phytoplankton that reached 108% and 170%, respectively. Rotifers may play a minor role in phytoplankton grazing. Preliminary microcosm and chemostat experiments have indicated that rotifers, due to their relatively low community grazing rates, should generally not be able (in contrast to some cladocerans) suppress the microbial web via grazing (Arndt, 1993). Grazing by zooplankton appeared to be more important in

determining changes of phytoplankton communities than seasonal physico-chemical variables.

5.5.2.2 Zooplankton

The composition and abundance of zooplankton among pools 14 and 3 and 6 were different. Large cladocerans such as *Daphnia* were most abundant in pools 3 and 6 and in contrast, smaller cladocerans such as *Bosmina* and rotifers were dominant in pool 14. Differences in zooplankton composition between pools 3, 6 and 14 could be explained by absence and presence of fish, identified as an important local factor (Hrbacek et al. 1962; Brooks and Dodson, 1965). This was consistent with Cottenie et al. (2003) who showed that local environmental determinants such as competition and predation can be strong enough to structure local zooplankton communities in lakes.

Pools 3 and 6 are fishless whereas pool 9 is a fish habitat. Fish can remove and control the population of *Daphnia,* which they are easily seen by their predators and move slowly through the water (Moss, 1997). Marchessault and Mazumder (1997) showed that *Daphnia* were virtually absent in enclosures with fish. In contrast, smaller zooplankters such as rotifers and copepods were present in high densities in a pond with fish because potentially competitive zooplankters such as *Daphnia* have been removed by fish. Copepods can rapidly move to avoid fish attacks and rotifers are too small a potential meal to justify the investment of an attack. This result was also consistent with Anderson et al. (1978) reporting that intense fish predation results in a predominance of smaller cladocerans (e.g. *Bosmina*, *Chydorus*) and rotifers.

A regional factor may also influence abundance of zooplankton populations. Decline of cladoceran populations in pool 14 in summer was possibly due to increased water temperature in early summer that resulted in greater consumption of zooplankton by planktivorous fish (Mills and Forney, 1981). In Lake Mendota, predation by perch and cisco was sufficient to keep populations of *Daphnia galeata* (G.O. Sars) at low densities during July and August in spite of more abundant algal food resources available, and even a slight increase in water temperature, e.g. due to global warming, substantially enhances the predation rate of fish on *Daphnia* (Luecke et al. 1990; Mehner, 2000).

Lack of submerged macrophytes could also be related to low densities of large cladocerans such as *Daphnia* in pool 14 that would mean no refuge for *Daphnia* to hide from their predators during the day (Scheffer, 1999). Moss et al. (1997) explained that by day, due to shading by the plants and with abundant plant beds, the fish may not see *Daphnia* well enough to risk an attack among the clutter of stem and leaves in the plant beds, or they may be deterred from making too many obvious and risky movements in a place where their own predators, large perch and pike may lurk. A study on spatial avoidance of littoral and pelagic invertebrate predators by *Daphnia* suggested that *Daphnia* moved into the vegetation in the presence of the pelagic *Chaoborus* to avoid predation (van de Meutter et al. 2005).

In contrast, abundance of vegetation in pools 3 and 6 may relate to high numbers of zooplankton as submerged plants protect zooplankton from predation (Timms and Moss, 1994). Leibold (1999) found a positive relationship between the densities of zooplankton and plants in fishless ponds and Blindow et al. (2000) reported that high densities of cladocerans were found in lakes within dense stands of Charophyta. Paterson (1993) also discovered large numbers of zooplankton among macrophyte communities. Shading of aquatic plants hides zooplankton and helps them not to be seen by predators; a complementary refuge may be created by the chemical and structural conditions in the plant beds, which may impede the feeding of some species of young fish (Moss et al. 1997).

High densities of large cladocerans such as *Daphnia pulex* (Straus) in pools 3 and 6 may result in relatively lower populations of rotifers compared with that in pool 14. Rotifer populations in pool 14 exponentially increased after the cladoceran population declined. Thus cladocerans may have an adverse effect on rotifer populations. In this case, large cladocerans such as *Daphnia* may suppress rotifer population by competing for food as they are bigger in size and more efficient feeders. In zooplankters, filtering rate is proportional to the square or to the cube of their body lengths and thus filtering rates increase with increasing body size (Burns, 1969). Brooks and Dodson (1965) found that large (1.3-3.0 mm) zooplankton species dominated over small (0.2-1.3 mm) ones in the absence of fish predation because they feed more efficiently. Neil (1984, 1985) also clearly illustrated that removal of

Daphnia from enclosures caused the rotifer biomass density to increase to about ten times that in control enclosures.

Mechanisms by which *Daphnia* can suppress rotifers include exploitative and interference competition. Gilbert (1988) explained that since all of the food resources utilised by rotifers are also utilised by *Daphnia*, rotifers cannot avoid exploitative competition with *Daphnia* and they may generally be at a disadvantage. The basis for the superior exploitative ability of the *Daphnia* is also likely due to a greater ability to withstand temporary starvation between periods of food addition, a lower threshold food level or both (Gilbert, 1985a). Large cladocerans such as *Daphnia* may also mechanically interfere with rotifers. Direct observations with several species of *Daphnia* and rotifers (*Keratella cochlearis* (Gosse)) by Gilbert and Stemberger (1985) and Burns and Gilbert, (1986b) revealed that interference occurs when the rotifers are swept into the brachial chamber of a feeding *Daphnia*. Some rotifers are ingested and many that are retained in the chamber for more than several seconds are rejected in a damaged condition and this damage may be lethal and sub-lethal. Interference competition also includes the detachment of eggs from ovigerous individual rotifers. Separated eggs may not hatch and are more susceptible to some predators (Gilbert, 1988).

5.5.2.3 Macroinvertebrates

Changes of macroinvertebrates associated with plants and in the sediment in all studied pools showed similar tendency in that population of macroinvertebrates increased in summer and decreased in winter. Thus, a regional factor was a main factor influencing seasonal variation and abundance of macro invertebrate populations. Temperature could be related to changes in macroinvertebrate populations. Leeper and Taylor (1998) found spring peaks of chironomid emergence at Rainbow Bay in South Carolina, U.S.A. and seasonal shifts in the timing of peak emergence and this may be related to variation in water temperature (Wrubleski and Rosenberg, 1990). When extensive areas of Rainbow Bay remained inundated, high rates of chironomid emergence would continue through the warm summer months.
Macrophytes appeared to influence abundance and structure of invertebrate communities. This was with accordance with Statzner and Higler (1986) and Sandin

and Johnson (2004) reporting that local scale variables such as vegetation and some chemical variables were most strongly associated with assemblages of macroinvertebrates. In this study, I found that differences in biomass and species of macroinvertebrates associated with plants and in the sediment in pools 3 and 6, and pool 14 could also be related to availability and plentifulness of food sources. Many species of macroinvertebrates including gastropods (*Planorbis carinatus* (Walker) and *Physa acuta* (Draparnaud)), *Gammarus*, Baetidae and *Chaoborus* were associated with plants in pools 3 and 6 and they were abundant. In contrast, they were hardly found in pool 14 where aquatic vegetation was less dense. Talbot and Ward (1987) found that the gastropod, *Potamopyrgus antipodarum* (J.E. Gray), was dominant in all major macrophyte communities in Lake Alexandrina. New Zealand. Lillie and Evrard, (1994) and van den Berg et al. (1997) also stated that high levels of food sources and availability in dense aquatic plant communities, and large area of macrophyte coverage contributed to high macroinvertebrate densities. Furthermore, Diehl (1992) noted that abundant vegetation increased the biomass of all macroinvertebrates, compared with that in treatments lacking plants.

Biomass, species composition and seasonal duration of aquatic plants all determine the composition and abundance of macroinvertebrates (Lodge, 1985; Rooke, 1986) since macroinvertebrates such as miners and shredders directly feed on macrophytes whereas grazing and scraping organisms consume periphyton (Rabe and Gibson, 1984; Newman, 1991). Other species of macroinvertebrates such as epiphyton grazers also occur on vegetation while benthic invertebrates abound in submerged plant beds (Moss, 1997; Vermaat, 1994). van den Berg et al. (1997) showed that seasonal variations in densities of the dominant mollusc species were closely related to the development of *Chara* biomass as well as to periphyton cover on charophytes and thus the relationships between the biomass of macrophytes and macroinvertebrates might be explained by considering food preferences of the organisms involved (Dvorak and Best, 1982).

Shallow pools 3 and 6 had higher macrophyte biomass compared in pool 14, which is deeper. The higher densities and greater percentage of coverage of aquatic plants may provide greater shelter for macroinvertebrates to hide themselves from predators such as fish or other carnivorous animals (Lillie and Evrard, 1994). Hanson (1990) and

Diehl (1992) showed that macrophytes such as charophytes in particular, act as refuges for macroinvertebrates to avoid fish predation or tended to delay the negative effect of fish on predatory invertebrates because invertebrates can hide themselves from predators. Diehl (1992) showed that in the absence of vegetation, fish like perch successfully increased their consumption of micro crustaceans. In contrast, in deeper pool 14, aquatic vegetation was sparse, thus limiting refuge for macroinvertebrates from predation (Crowder and Cooper, 1982).

The presence or absence of fish could also be related to lower and higher biomass of macroinvertebrates in the pools. This study clearly showed that higher biomass of invertebrates was found in pools 3 and 6 without fish. In contrast, pool 14 had the lowest values of invertebrate biomass. Fish may have significant effects on the biomass, abundance and biodiversity of macro invertebrates (Andersson et al. 1978; Power, 1990; Rosenfeld, 2000). Several studies have demonstrated that abundance of larger invertebrates decreased in the presence of fish and effects of fish predation have been demonstrated much more frequently for large and mobile benthic invertebrates than for small or cryptic taxa (Macan, 1966; Morin, 1984; Diehl, 1992). This is likely because many species of invertebrates are the main diet for fish and this is confirmed by studies on gut contents of wetland fish such as brown bullhead (*Ictalurus nebulosus* (Lesueur)), black crappie (*Pomoxis nigromaculatus* (Lesueur in Cuvier and Valenciennes)) and common carp (*Cyprinus carpio* (Linnaeus)) revealing large numbers of midge larvae (Diptera: Chironomidae) (Batzer, 2000).

Although pools 3 and 6 are fish-free habitats, total numbers of cladocerans in those pools were still lower than in pool 14 (*Figure 5.16*). Therefore, there may be other invertebrate predators that control the population of cladocerans in pools 3 and 6. I found that *Chaborus* were more abundant in pools 3 and 6 and less in pool 14 (*Figure 5.16*) and this was likely due to the absence of fish in pools 3 and 6 that promoted alternative predators such invertebrate to flourish. This was in agreement with Nyberg (1998) and Irfanullah and Moss (2005) who found that in fishless lakes, invertebrates like phantom midge larvae (Chaoborus spp.) may increase in abundance, and may shape the zooplankton community as top predators. *Chaoborus* preys heavily on both *Daphnia* and smaller cladocerans especially those with intermediate sizes between 0.8-1.0 mm (Dodson, 1974; Havel and Dodson, 1984; Pastorok, 1980; Spitze, 1985).

Other invertebrate predators such as dragonfly nymphs (*Epitheca cynosure* (Say)) and damselfly larva (*Ischnura* sp.) also can eliminate *Daphnia* (Burks et al. 2001; van de Meutter et al. 2005).

5.5.2.4 Macrophytes

Local factors such as pool morphology, fish and birds seemed to be crucial in structuring macrophyte communities in the studied pools. Species and numbers of macrophytes were different in deep and shallow pools. Plants were more diverse and present in higher numbers in the bigger pool 6 and fewer in the smaller pools 3 and 14. This finding was similar to other studies in that species richness of submerged macrophytes increased markedly with lake size (Amarasinghe and Welcomme, 2002; Bazzanti et al. 2003; Sondergaard et al. 2005) and this is likely because of greater areas for plants to colonise and distribute across the lake.

Water depth could also be related to abundance and biodiversity of aquatic vegetation. Only a few species of macrophytes were present in the deeper pond 14. Kirk (1984), Havens (2004), Coops et al. (2007) and Xiao et al. (2007) suggest that water depth is a major environmental factor influencing the distribution and growth of submerged macrophytes since macrophytes require energy in the form of solar radiation for growth. Xiao et al. (2007) showed that water depth significantly affected clonal growth of *Vallisneria natans* (Lour.) in terms of clone weight, number of generations and stolon length. Interestingly, Middelboe and Markager (1997) discovered that different species of plants occur at different depths in lakes. For examples, in lakes with transparency as secchi disc depth less than 7 m, caulescent angiosperms and charophytes penetrated deepest followed by bryophytes and *Isoetes* spp. Furthermore, rosette-type angiosperms had the lowest colonisation depth in all types of lakes.

Drying out and rewetting may also be another important factor affecting aquatic plant communities especially in shallow ponds. Several studies have shown that temporal declines in water level may enhance macrophyte abundance (Havens et al. 2004) as disturbances caused by drawdown may prevent competitive dominance, thereby increasing species richness (Hill et al. 1998). Another possible reason is because

drying out of highly eutrophic pools produces fertile land and promotes colonization of other types of terrestrial and marginal plants.

An increase of water levels may affect abundance and species richness of macrophytes too. For example in 2007, water levels in all pools had continuously increased and remained high. A survey of macrophytes found that numbers of plant species in pool 3 had declined and some species of submerged plants such as *Potamogeton* had disappeared. This is likely because increasing depth may decrease light availability and may impact species richness and abundance of under water macrophytes. This is in agreement with Coops et al. (2003) who reported that high water levels may limit submerged plant expansion including a shift to a sparsely vegetated state, whereas a substantial reduction of lake level may encourage expansion of submerged plants.

Presence of fish such as carp in pool 14 is also likely to influence macrophyte communities. There is evidence that fish have considerable effects on vegetation and reduce biomass of submerged macrophytes (Johansson and Persson, 1983). Several studies have shown that the common carp (*Cyprinus carpio* may destroy submerged vegetation by eating and uprooting plants (King and Hunt, 1967; Crivelli, 1983) or they may cause the disappearance of vegetation indirectly by increasing the turbidity (Moyle and Kuehn, 1964). Another species of fish destructive to aquatic macrophytes is rudd (*Scardinius erythrophthalmus* (Linnaeus)). In Lake Zwemlust, the Netherlands, about 40% of the maximum macrophyte biomass in 1991 was estimated to be consumed by rudd (van Donk et al. 1994) Therefore, fish grazing on macrophytes may affect the internal balance among autotrophic components by reducing the biomass of macrophytes, thereby reducing their ability to compete with algae for nutrients and turning a lake to a turbid stage that is macrophyte-free (Hansson et al. 1987).

Invasive floating plant species such as *Azolla filliculoides* appeared to cause a negative problem in pool 3 as density of such floating species could be related to the decrease and absence of submerged macrophytes. In 2007, I found that floating plants, *Lemna trisulca, Lemna* spp. and *Azolla filiculoides,* covered nearly all of water surface area of the pool and were mostly within about 5 cm from the surface. And in

this year, *Potamogeton* spp. and other submerged macrophytes species were not found (*Figure 5.34*). In July 2001, Whild and Lockton also reported that water fern, *Azolla filliculoides*, entirely covered the surface of this pool. The appearance of water fern could be due to deliberate introduction (Whild and Lockton, 2003). *Azolla* is widely stocked in gardens.

Figure 5.34 Floating plants covered the entire surface of pool 3 in 2007

Thick mats of floating species are likely to be a main cause of disappearance of submerged plants due to reduced penetration of light to the underwater environment. Janes et al. (1996) and Sinitro et al. (2006) found that floating macrophytes caused light deficiency and physio-chemical changes and these have an adverse and direct impact on submerged plant communities. Janes et al. (1996) reported that *Elodea nuttallii* (Planch.) and *Potamogeton crispus* (Linnaeus) showed reduced branching and biomass production beneath dense mats of *Azolla filiculoides*. Scheffer et al. (2003) suggested that lakes that are deeper and occupied by dense mats of free floating plants have a lower probability of being dominated by submerged vegetation as a result of the shading effects of floating plants on submerged plants. They also stated that invasion by floating plants threatens the functioning and biodiversity of freshwater ecosystems because dark, anoxic conditions under a thick floating macrophyte cover leave little opportunity for underwater animal and plant life.

Crassula helmsii, known as New Zealand Pygmy weed, and found in pool 6 is also considered a cause for concern at Brown Moss. It was spread around the margins and

within the pool and this may negatively affect native species. Dawson, (1988, 1989) and Huckel (2002) stated that *C. helmsii* causes major environmental problems in freshwater aquatic habitats as it has spread rapidly and widely throughout the UK. It forms dense vegetation mats that choke ponds and drainage ditches. In smaller water bodies it can rapidly become dominant over other vegetation and it is thought that *C. helmsii* is out-competing plants at some sites, reducing biodiversity (Langdon et al. 2003). An evidence suggests that through competition *C. helmsii* is capable of reducing the diversity of plants at pond margins (Leach and Dawson, 1999). Prevention of its further spread, especially in the pool and around the site is urgently needed.

5.5.2.5 Birds

A combination of regional and local factors influenced bird populations whilst the presence of birds determined limnological characteristics of specific pools. The population of water birds in pools 3, 6 and 14 changed seasonally throughout the year as elsewhere (Kitchell et al. 1999; Noordhuis et al. 2002). Aquatic birds in pools 6 and 14 were abundant and diverse in winter and fewer in summer. Disappearance of waterfowl and decrease in numbers in summer may be likely due to the fact that they moved to other water bodies. However, some species such as teal are migratory and move northwards in summer. In contrast, in pool 3, the bird populations increased in summer 2005 perhaps due to local movement from pool 6, which had dried out.

Lake morphology appeared to influence species and numbers of aquatic birds. There were more species and numbers of aquatic birds in pool 6 than in pools 3 and 14. This is probably because the surface area of pool 6 is larger than that of other pools. Bigger birds such as Canada geese, and swans in particular, were confined to pool 6. Since trees around pool 6 have also been removed, this created more open space around the pool for large aquatic birds to flight in and take off. Compared with pool 6, pools 3 and 14 are smaller and surrounded by mature and high trees and this may cause difficulty in landing for such big birds. A study of the habitat use of aquatic bird species by Des Granges and Darveau (1985) revealed that waterfowl are seen more frequently on certain types of lake, characterized by a level of acidity, morphology and altitude. They found that common loon, *Gavia immer* (Brunnich), primarily use

large, low altitude lakes, whereas the ring-necked duck *Aythya collaris* (Donovan) and the American black duck are often found on smaller, higher altitude lakes. Suter (1994) also stated that given the large size range of 20 lakes studied, a high correlation between size and bird species numbers is to be expected. Furthermore, suitability for foraging such as shallow areas and percentage of natural, open shoreline are among other variables affecting abundance and species richness of waterfowl in lakes (Suter, 1994).

High numbers of birds in pool 6 were also likely due to availability and plentifulness of food sources. Since the pool dried up in 2005, more species and numbers of terrestrial and amphibious plants such as *Alopecurus aequalis, Atriplex patula* and *Oenathe aquatica* occupied the pool and become abundant. And when the pool was underwater in the following winter, these plants turned into food sources together with newly germinated submerged plants such as *Nitella* and filamentous algae that attracted more birds to feed on them. Hoyer and Canfield, Jr. (1994) and Brown and Dinsmore (1986) support this finding. They stated that productive aquatic ecosystems are able to support a greater number and biomass of organisms and more specialised species and macrophytes, particularly, are important for bird populations. In breeding situations, Suter (1994) found that food availability and the richness of the shoreline vegetation were important. In this study more chicks were seen in pool 6 rather than in pools 3 and 4.

Furthermore, feeding by people visiting pool 6 may attract more waterfowl and keep them around the pool (*Figure 5.35*). In summer 2005, there were mallard around pool 6 although the pool had completely dried out. These remaining birds may be attracted by people feeding them. Bock and Lepthien (1976) showed that between 1962 and 1971 the North American blue jay (*Cyanocitta cristata* (Linnaeus)) population increased by 30% and that migration of this species declined and the most likely cause was an increase in winter feeding by people. Winter feeding might also change the distribution pattern of some species. In addition, bird feeding may cause negative impacts on water chemistry and environmental and wildlife problems. Food such as bread thrown for birds that is left over may lower water quality. Feeding birds could also be detrimental to their health, as it is not their natural food that is given. At St. Saviours Reservoir in States of Guernsey and elsewhere, signs are placed around the

reservoir, reminding the public not to feed the wildlife in the area (Guernsey Water, 2007).

Figure 5.35 Bird feeding at Brown Moss

High numbers of birds gathering in lakes are likely to affect the aquatic ecosystem, particularly aquatic macrophytes (See more in Chapter 6). Because many birds feed on macrophytes, their feeding and trampling may lead to destruction and losses of aquatic plant communities and of biological diversity. In winter 2005, I found that herbivorous waterfowl mainly teal and mallard grazed heavily on vegetation across the lake (*Figure 5.36*). This result was consistent with Klaassen and Nolet (2007) who found that water plants may be crucial for water birds during periods of high energy requirement such as migration. Lauridsen et al. (1993) investigated the impact of waterfowl grazing on colonization of submerged macrophytes in shallow Lake Væng and found that water birds controlled and delayed macrophyte colonization, following removal of fish. Figure 36 shows severe damage to *Oenanthe aquatica* (left) and physical damage of *Juncus effusus* caused by trampling (right). Feeding and trampling can damage stems, leaves and other parts of plants and may change species composition as many species may disappear if they are susceptible.

Figure 5.36 Physical damages of macrophytes in pool 6, caused by feeding and trampling of aquatic birds

Apart from feeding and trampling effects on aquatic macrophytes, addition of nutrients by waterfowl may also lower water quality, especially when they gather at high densities. Canada geese are introduced species in the United Kingdom but nuisance problems from them are also reported in the eastern United States. Conover and Chasko (1985) revealed that droppings from large flocks of geese can contribute to the over-fertilization of small lakes and reservoirs in Connecticut and Jefferies (2000) and Vanni (2002) found that high densities of waterfowl have the potential to cause increased nutrient loading and subsequent changes to water quality in a wetland and further downstream. As bird droppings contain nitrogen and phosphorus, such nutrients can stimulate growth of phytoplankton and cause eutrophication problems (Marion et al. 1994; Post el al. 1998; Manny et al. 1994). Marion et al. (1994) also reported that local concentrations of phosphorus on areas directly occupied by colonies of herons were 42 times greater than outside the colonies. Furthermore, they stated that during the plant growing period between April and September, the contribution of birds can increase to 37% of total P input of the lake they studied. In addition, droppings of birds can cause other problems in lakes such as depletion of

dissolved oxygen. Brinkhurst and Walsh (1967) reported that the result of guanotrophication of Rostherne Mere, in Cheshire, England was oxygen depletion in deeper parts resulting from the excessive amounts of droppings deposited by roosting gulls. However, they were mistaken (Carvalho, 1993).

In conclusion, shallow pools 3 and 6 were hypereutrophic and macrophytes dominated whereas hypereutrophic pool 14 is dominated by phytoplankton. I found that there were several factors affecting limnological characteristics in the studied pools and some pools at Brown Moss are suffering from the effects of cultural eutrophication and invasive species.

Pool 3 is less disturbed by birds and agricultural activities compared with pools 6 and 14. High concentrations of phosphorus originally of allochthonous origin are being cycled within the pool through sediment release and uptake, and nitrogen is scarce. Macrophytes were present in high numbers in the pool and they supported communities of aquatic macroinvertebrates and were shelters for zooplankton. Changes of water levels and invasive species and dense floating plants are likely to affect abundance and species richness of submerged macrophyte communities. In a fishless habitat, zooplankton was abundant and *Daphnia* effectively controlled populations of phytoplankton although concentrations of nutrients were quite high. In the absence of fish, alternative predators may potentially become an important factor structuring zooplankton in the pool.

In pool 6, main sources of nutrients come from birds and the sediment. Aquatic birds were present in high densities in the pool, especially during winter. Birds were attracted to the pool by plentifulness of food sources and people feeding them. Waterfowl, especially introduced species such as Canada geese, increased nutrients and lowered water quality whereas other species such as teal and mallard damaged aquatic vegetation across the lake. Shallowness and drying out of the pool promoted colonisation and abundance and richness of macrophyte species. However, drying out of the pool killed fish and survivors were removed. With the absence of fish and richness in macrophytes, zooplankton and macroinvertebrates were present in high numbers. Macrophytes played a crucial role as food sources and refuges for aquatic fauna and appeared to help maintain a clear state of the pool at least in summer.

Abundant cladocerans, particularly *Daphnia* prevented growth of phytoplankton and kept the lake clear, although the pool is categorised as hypereutrophic lake. *Daphnia* also appeared to influence the abundance of smaller crustaceans such as rotifers, and or copepods. Further research about distribution and the potential effects of *Crassula helmsii* on the site should be carried out in order to control and prevent problems caused by such species.

Excessive amounts of nutrient inputs in pool 14 stem mainly from the arable agricultural catchment and grazed pastures following heavy rainfall. Horses in grazed pastures appeared to disturb colonisation of marginal plants of the pool and to cause erosion and soil compaction. A buffer zone is recommended to reduce the impact of nutrient enrichment. Pool 14 is relatively deeper with steep edges compared with pools 3 and 6 and this limited colonisation of marginal and submerged plants. Poor development of macrophytes was also likely due to effects of bird and herbivorous fish in the pool. Presence of fish and lack of submerged macrophytes in pool 14 resulted in low numbers of macroinvertebrates, scarcity of big cladocerans such as *Daphnia*, and abundance of smaller crustaceans like rotifers and copepods. With the absence of *Daphnia*, the pool is phytoplankton dominated. Major features of comparative limnology of the study pools are presented in Figure 5.37.

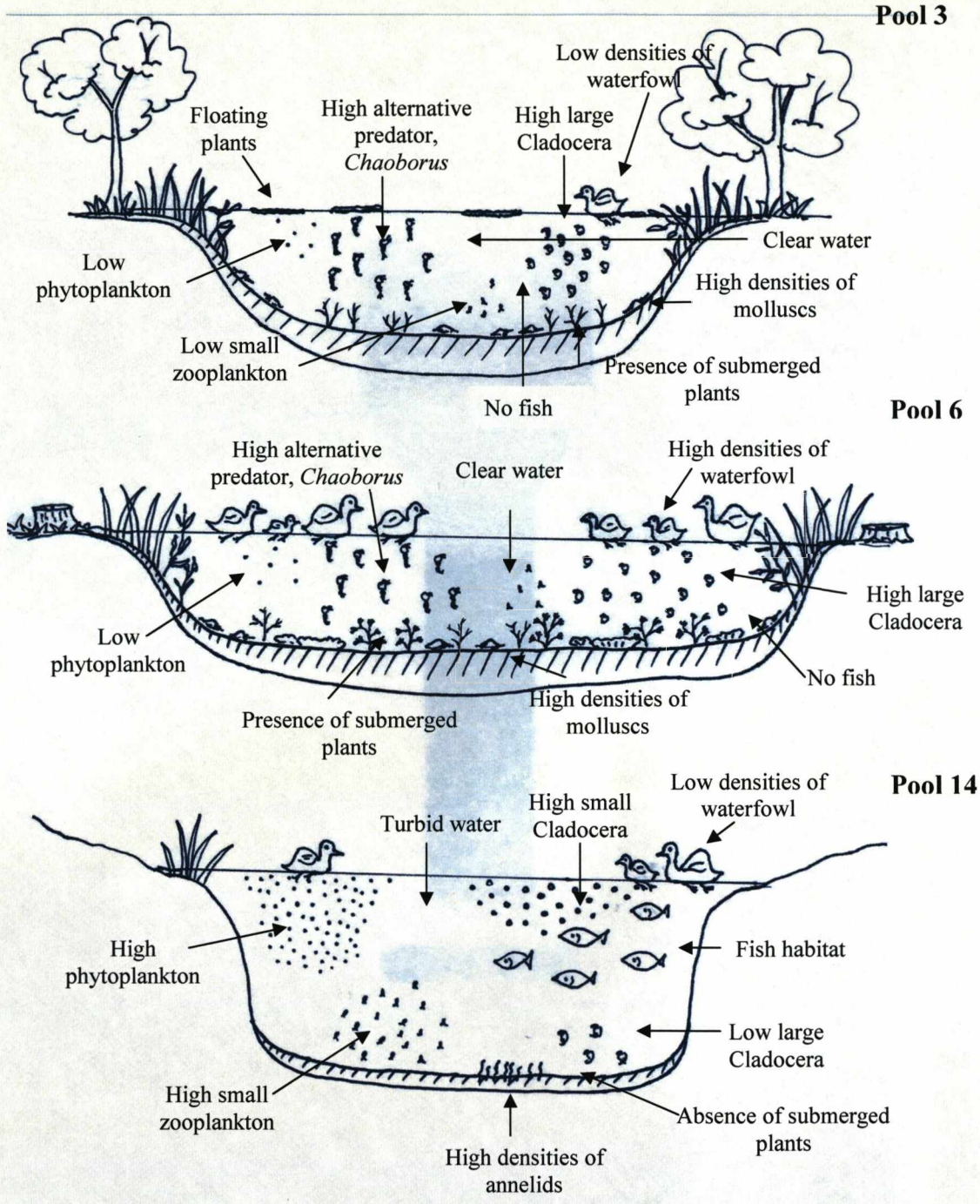


Figure 5.37 Comparative study of the limnology of pools 3, 6 and 14

Lastly, conditions and differences in adjacent pools were determined by both regional and local factors. It was clear that regional factors (the weather and landscape) strongly influenced physical and chemical properties of the pools such as changes in water levels and temperature. A local factor such as the sediment that was a source of internal loading also impacted water chemistry in the pools. In term of biological variables, regional determinants were important and impacted all the pools but mainly

on seasonal variations of biological communities. However, they were not involved in structuring the pool ecosystems. In fact, local variables such as biological interactions between phytoplankton and fish or between fish and zooplankton were the main cause structuring and determining biological communities in specific pools and making them very different.

Chapter 6 The impact of waterfowl on aquatic plants

Chapter 6
The impact of waterfowl on aquatic plants

6.1 Introduction

Submerged plants in shallow lakes play many crucial roles. Ultimately, they are food sources for animals such as invertebrates, fish and birds and sometimes for human beings. Aquatic macrophytes also serve as natural habitats as they provide sites for egg-laying, material for case-building and refuges for underwater animals such as zooplankton and small invertebrates (Odum et al. 1984; Engel, 1990; Williams et al. 1999; Kendra et al. 2000; Klaus et al. 2006). Plant beds harbour small animals such as cladocerans from fish predation in the day and at night the animals drift out of the beds and graze on phytoplankton (Timms and Moss, 1984). Macrophytes also support biodiversity of fauna and have a high number of associated macro invertebrate species including detritivores, periphyton-scrapers and omnivores together with predatory carnivores (Dvorak and Best, 1982).

Furthermore, aquatic macrophytes provide oxygen as a product of photosynthesis. Oxygenating macrophytes may temporarily increase oxygen levels in water bodies and even the roots of some emergent plants are used by specialist benthic animals to obtain their oxygen by piercing the roots (Williams et al. 1999). Aquatic plants in the littoral zone especially also soften the line of hard reinforced banks and provide a natural buffer between a developed shoreline and the open water. They also stabilise bottom sediment and protect shorelines from erosion and collapse.

Several studies have also shown that aquatic plants possess an outstanding ability for assimilating nutrients, maintaining water clarity and creating favourable conditions for microbial decomposition of organic matter (Balls et al. 1989; Brix and Schierup, 1986; Blindlow et al. 1993; Moss et al. 1996). They may be a sink for available nutrients in both water column and sediment and may maintain a competitive advantage over algae through luxury uptake (Ozimek et al. 1990; van Donk et al. 1993). In natural ponds or treatment systems, many species of aquatic plants including floating plants (water hyacinth (*Eichhornia* spp.)) or emergent plants (the common

reed (*Phragmites australis* (Cav.)) help purify contaminated water or absorb heavy metals (Boyd, 1970; Brix and Schierup, 1986).

Besides ecological significance, aquatic macrophytes are of interest in terms of scientific research and aesthetic values. A large number of studies have paid attention to aquatic vegetation and have covered various aspects ranging from their growth, production and physiology to interaction of plants with the aquatic ecosystem (Jeppesen et al. 1997; Egbert et al. 1999). Furthermore, plants can provide lush views and interesting reflections (Williams et al. 1999), and the natural beauty of aquatic vegetation, such as flowers and emergent and floating-leafed plants, is valued for its aesthetic qualities and inspirational for artists.

Even though submerged macrophytes are valuable in aquatic ecosystems, they are frequently lost from shallow waters. One factor affecting temporary and permanent loss and damage is aquatic birds. In ecosystems, aquatic birds are important and play several roles. For example, they control pests and insects on farms and provide fertilizers for the plants and lakes. Brandvold et al. (1976) showed that bird droppings modified the organic and mineral contents of water in ponds. Also, water birds have long been considered major dispersers of aquatic organisms, transporting plants and invertebrate propagules in their guts (endozoochory or internal dispersal) or attached to their bodies (ectozoochory or external dispersal) (Darwin, 1859; Ridley, 1930). In addition, some seeds consumed by bird have shown improved germination after consumption and defaecation (Agami and Waisel, 1986). Water birds such as kingfishers, cormorants and herons also have roles in maintaining fish populations at levels which can co-exist with and not threaten the system (Winfield, 1990; Moss et al. 1997). Thus, many preserved areas are conserved especially for bird communities as such birds are important and many are threatened.

However, birds can be problems in nature reserves and recreational areas, particularly when they become too abundant owing to feeding of them by the public or when they are introduced. Gathering of large flocks of water birds is considered a significant cause of the reduction of aquatic macrophytes where they are main food sources of birds (van Donk et al. 1996; Marklund et al. 2002). Overgrazing and trampling may result in low development of aquatic plant communities and may lead to a temporary

or permanent loss of vegetation (Strand, 1999). When aquatic macrophytes are absent from lakes due to over grazing, lakes lose algal inhibitors and thus herbivorous waterfowl may trigger a change from the clear-water, macrophyte-dominated state to the turbid state when nutrients are not sufficiently low to prevent algal blooms (Moss, 1990; Lauridsen et al. 1993; Mitchell, 1994; van Donk and Gulati, 1995; Sondergaard et al. 1996).

In addition, much human and financial effort has been directed at the restoration of shallow lakes in many areas to make lakes clear water-dominated by macrophytes and to preserve their high ecological and sociological values (Moss, 1990, 1997; Broads Authority, 1994). Establishing a stable aquatic vegetation is considered a very important tool to restore eutrophic lakes and reduce turbidity in shallow lakes (Timms and Moss, 1984; Scheffer, 1998; Schriver et al. 1995). However, the restoration process, usually through biomanipulation, may be slow or unsuccessful if grazing pressure of birds on vegetation occurs. Several studies have documented that grazing by herbivorous waterfowl may prevent re-establishment of submerged macrophytes and delay the recovery of shallow lakes undergoing restoration (Perrow et al. 1997; Lauridsen et al. 1993).

Understanding roles of waterfowl on macrophytes in a shallow lake is considered important since such knowledge may be useful to the management and reduction of impact of herbivorous aquatic birds on plants. Therefore, this study aimed to investigate seasonal interactions between water birds and aquatic macrophytes in a shallow lake. I also determined the grazing effects of herbivorous aquatic birds on vegetation by comparing the growth of plants in enclosures and in unprotected areas where birds had access. Lastly, I investigated whether waterfowl caused long term widespread loss and reduction of macrophyte biodiversity in a shallow lake that is also suffering from hydrological and nutrient problems.

6.2 Methods

6.2.1 Enclosure experiment

A field experiment was set up in pool 6 located in Brown Moss, from June 2005 until November 2007. The water surface area of this pool is 29,700 m^2 and the mean depth is 30.8 cm. In this experiment, I installed eight bird-proof enclosures and eight open water control treatments in random order in two separate shallow areas A and B of this pool (*Figure 6.1*). Eight plots (four enclosures and four open areas) were placed in the largest sub-area (A), which most birds inhabit and the other eight plots (four enclosures and four open areas) were installed in a small pool named B, usually connected to the biggest pool (*Figure 6.1*). The space between each treatment replicate was one metre. Enclosures were made of wire netting, including a top cover above water level with a mesh size of 25 mm. to prevent birds flying in. Each of them was 0.7 m in height. Open water treatments were marked by wooden sticks at the corners. .

Figure 6.1 Enclosure experiment set up in two areas, A (above) and B (below) in the largest pool (6) at Brown Moss in 2005

The aquatic vegetation in the treatments was monitored visually every six weeks, using a 1×1m quadrat. Plants were identified to species level and the biomass was estimated by volume occupied (proportion of volume infested (PVI)). I measured height (cm) and area coverage (m^2) of each species and the area estimated was then multiplied by the mean height of the macrophyte mass for each species to give a PVI value (McKee et al. 2002). PVI values of submerged, emergent and floating plants in all treatments were expressed a percentage of the volume of a bird cage (2×6×0.7 or 8.4 m^3). I also measured water levels using a 1-metre rule at the studied sites each time when I monitored aquatic plants. Bird numbers were monitored at 13.30 hr by methods given in Chapter 3 on days when plants were estimated.

I used SPSS 15.0 for one-way ANOVA analysis to compare differences in mean PVI values and numbers of plant species among control and enclosure treatments and among treatments and season. I also applied Pearson correlation coefficients (r) to determine correlations between numbers of birds and mean PVI values in both control and enclosure treatments.

6.3 Results

6.3.1 Enclosure experiment

Water availability at Brown Moss in 2005 was critical since water levels decreased dramatically. In late July 2005, water in areas A and B completely dried out as a result of lack of rain and high evaporation (*Figure 6.2*). After November 2005 the areas A and B received water from winter rain and both areas were entirely underwater again. Water levels at the beginning of 2006 remained high and decreased in summer. In 2007, water levels increased continuously and reached a peak of approximately 97 cm in July.

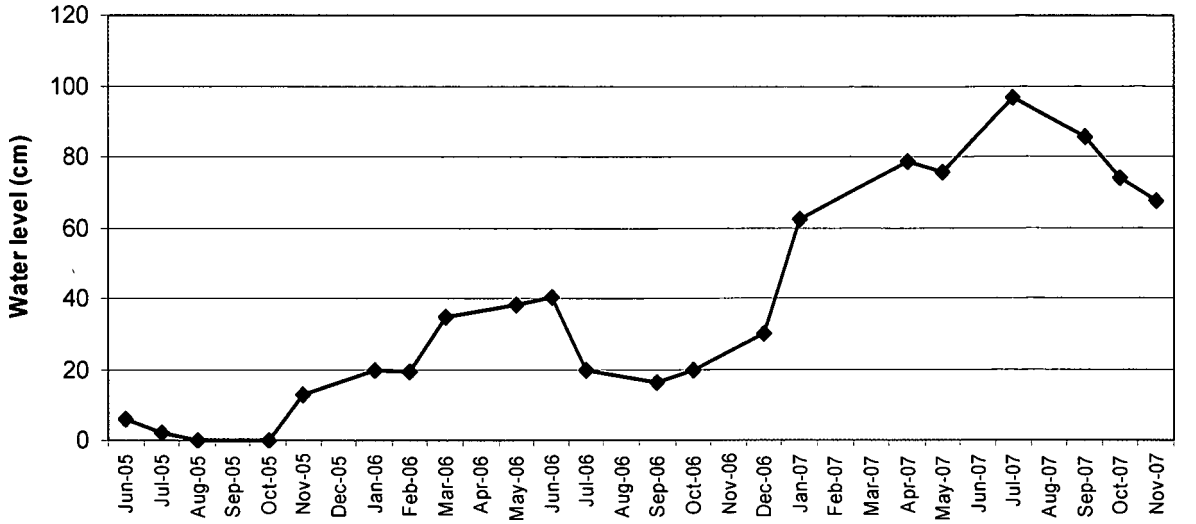


Figure 6.2 Variations of water depth (cm.) in the studied areas

Before water levels fell, there were a few species of submerged plants recorded in both studied areas A and B, but they were not abundant. As a result, total macrophyte PVI values of submerged plants in summer 2005 in both A and B were low. Aquatic plants recorded in area A between June and July 2004 were *Polygonum amphibium* and *Spirogyra* sp. whereas *Nitella* sp., *Polygonum amphibium* and *Spirogyra* sp. were present in area B (*Figure 6.3*).

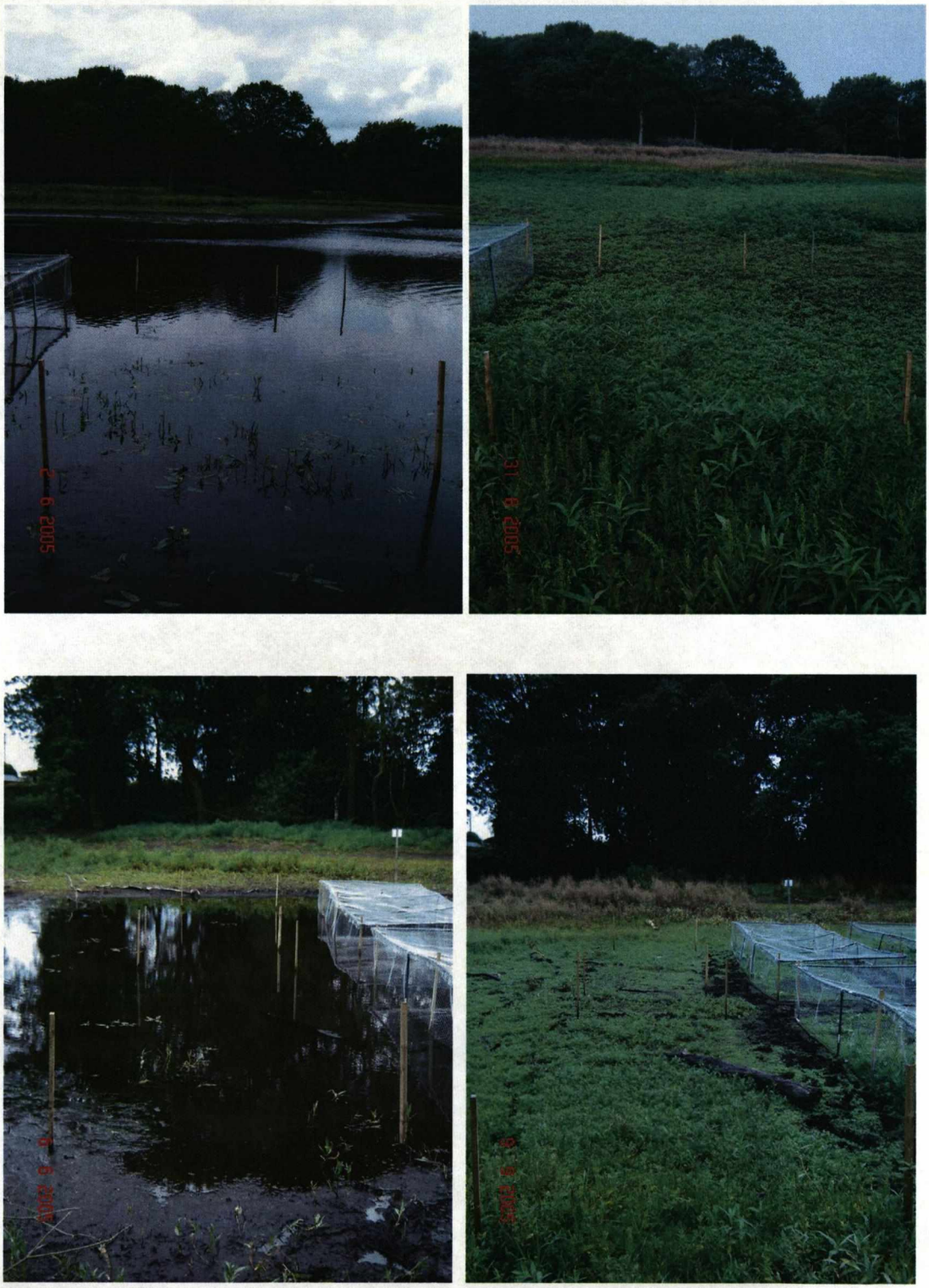


Figure 6.3 Enclosure experiments in A (above) and B (below) before (left) and after (right) water dried out in 2005.

When areas A and B became dry, terrestrial and amphibious plants such as *Alopecurus aequalis*, *Atriplex patula*, *Alisma plantago-aquatica*, *Oenanthe aquatica*, *Carex pseudocyperus* and *Epilobium ciliatum* had replaced submerged plants (*Figure 6.3*). *Alopecurus aequalis* and *Atriplex patula* appeared to be dominant in both areas A and B. These plants grew very well and were present in every treatment both in enclosures and in open areas. Since terrestrial plants had replaced aquatic macrophytes, PVI values of enclosure and control treatments area in A and B increased and were higher than those measured for submerged plots in summer 2005.

Overall, total PVI values of terrestrial plants in both enclosure and control treatments in areas A and B showed a similar tendency to increase in summers 2005 and gradually declined during the winters (*Figure 6.4*). During 2005 – 2006, total PVI values for macrophytes in enclosure treatments were relatively higher than those in open areas and the gap between total PVI in enclosures and open areas widened after winter 2005 and remained wide until autumn 2006. In spring 2006, total PVI values of both open areas and enclosures increased and were highest in June. In contrast, in winter 2006, total PVI values in control areas were higher than in enclosure treatments until late summer 2007 when total PVI values in enclosure treatments started to increase. Afterwards, both total PVI values in enclosure and control areas declined toward winter.

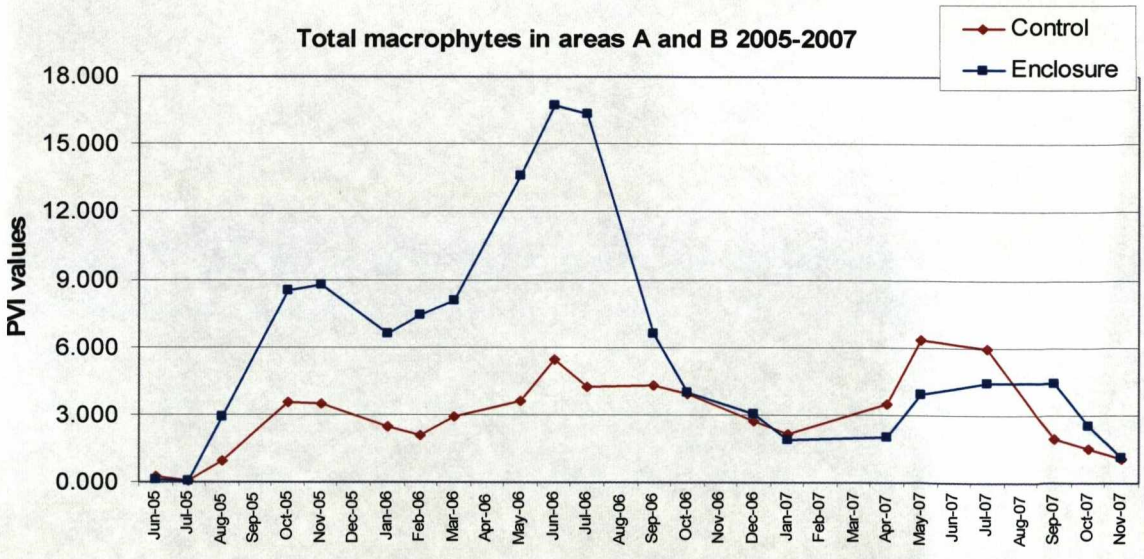


Figure 6.4 Seasonal variation of total PVI values for macrophytes in enclosure and control treatments from 2005-2007

Figure 6.5 shows total PVI values of submerged, emergent and floating plants in areas A and B. Submerged plants were more abundant in control areas. In contrast, total PVI values of emergent and floating plants in enclosure treatments were greater than in control plots. Submerged plants were over all less abundant than emergent plants.

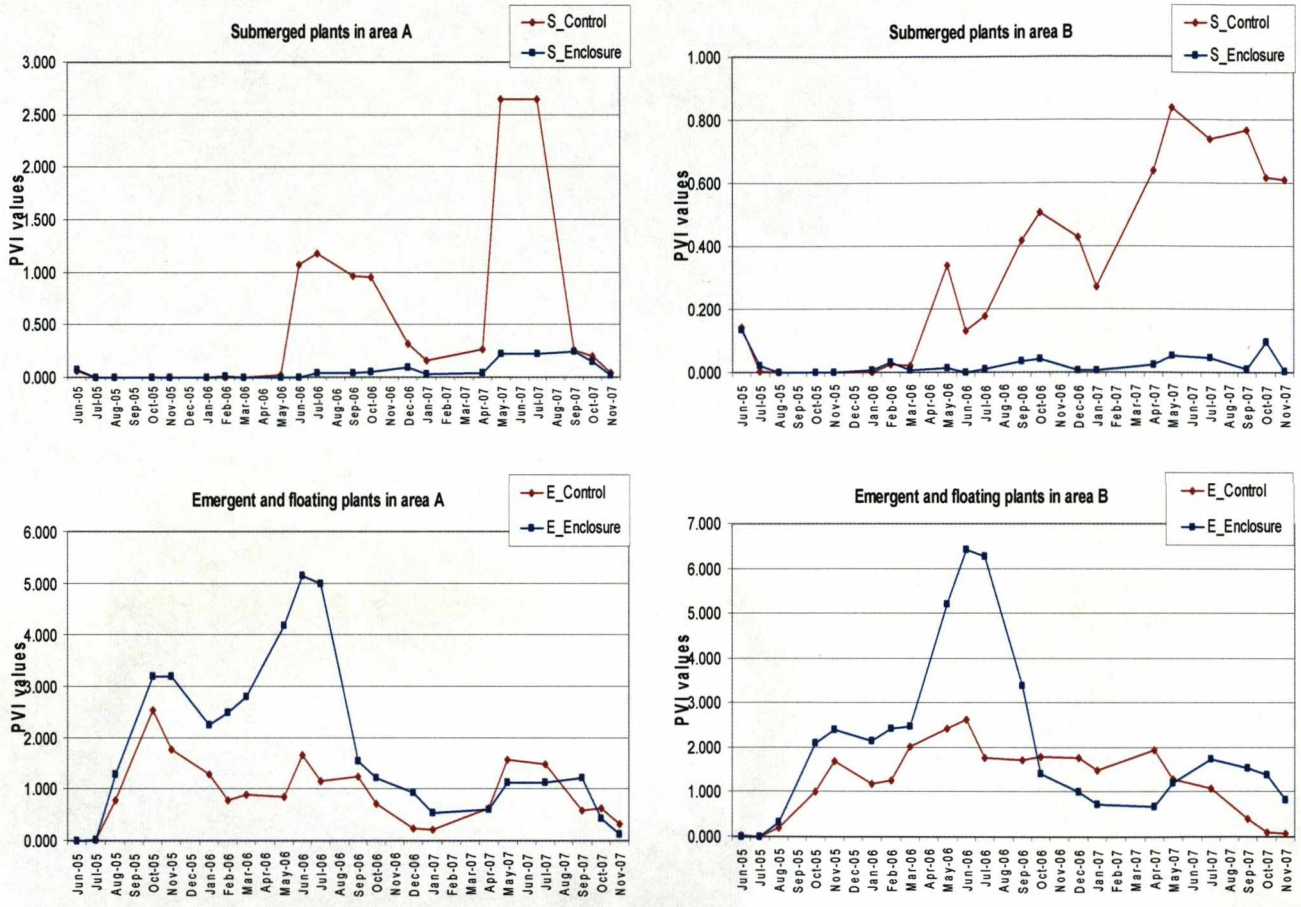


Figure 6.5 Total PVI values of submerged, and emergent and floating plants between protected and unprotected areas in A and B

The results for growth of individual main emergent species show that most of them present in the enclosures grew better than in open areas (*Figures 6.6 and 6.7*). Statistical analysis revealed that mean PVI values of macrophytes such as *Alisma plantago-aquatica* (F = 6.643, P = 0.017), *Juncus* spp. (F = 29.54, $P < 0.001$), *Oenanthe aquatica* (F = 9.952, P = 0.003), and *Typha latifolia* (F = 9.793, P = 0.004) in enclosure treatments were significantly higher than those in control treatments. However, there was no significant difference in mean PVI values of *Alopecurus aequalis* between control and enclosure areas (F = 1.395, P = 0.246).

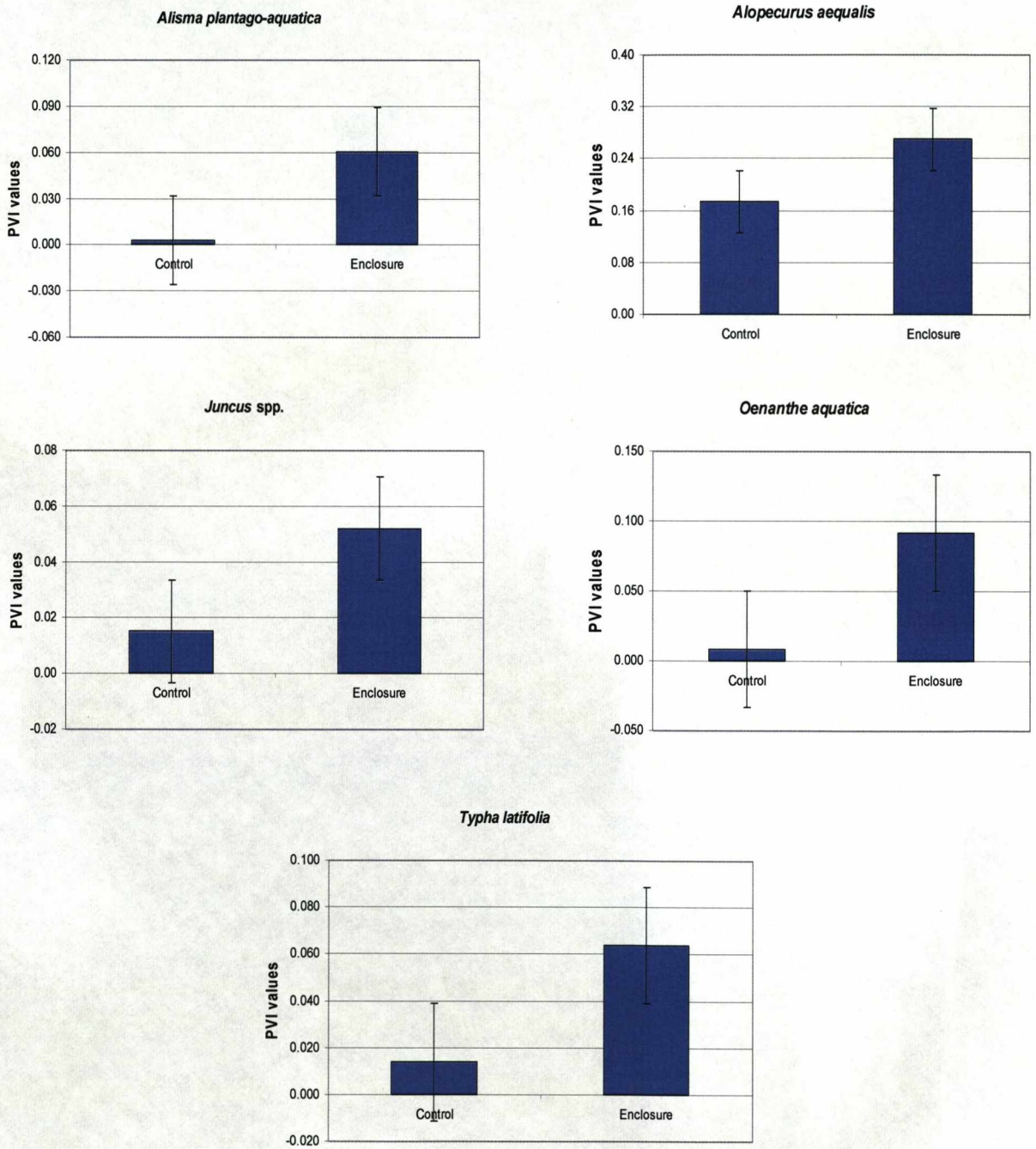


Figure 6.6 Comparison of mean PVI of plants in controls and enclosures (n = 2)

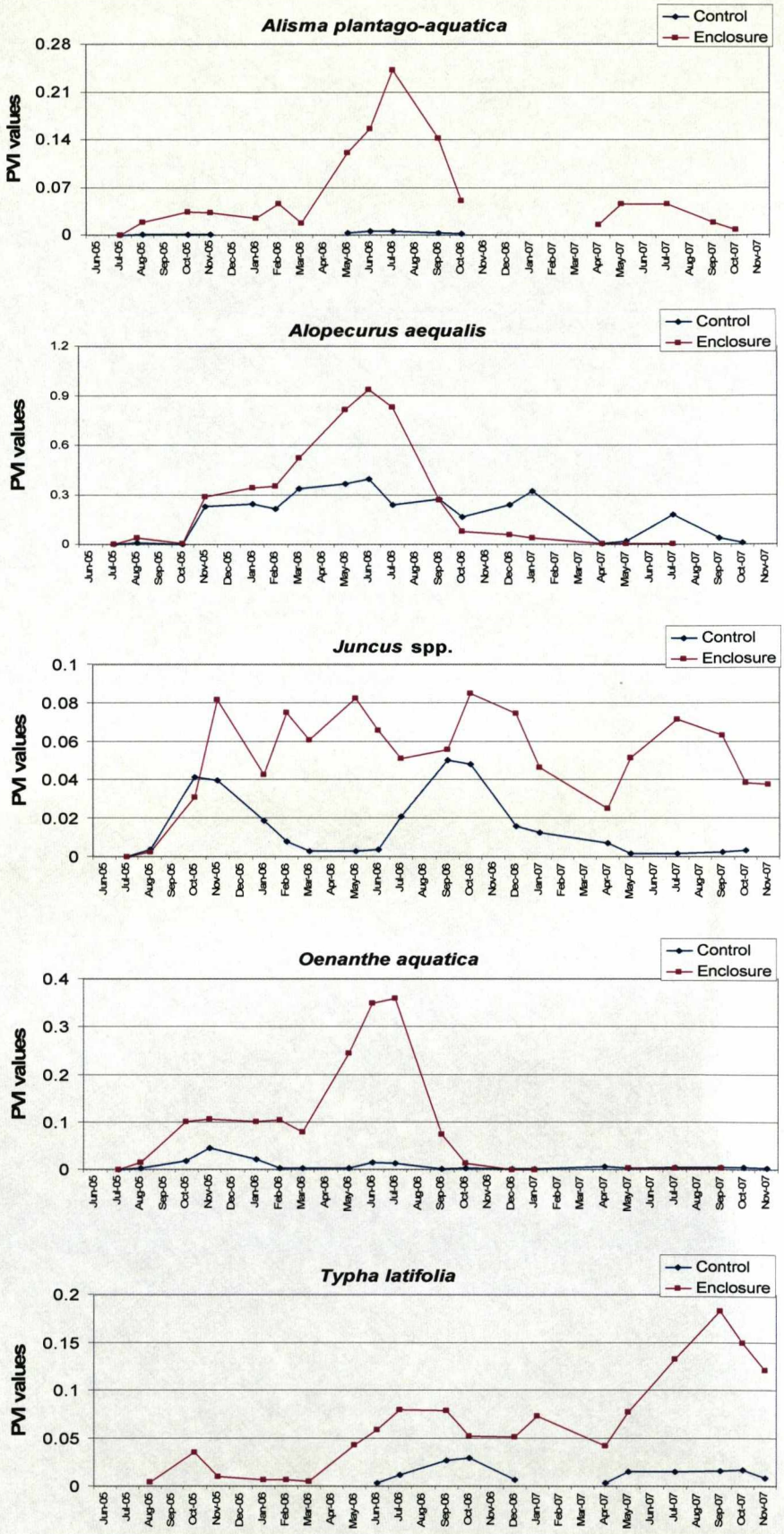


Figure 6.7 Seasonal growth of plants in the experiment between control and enclosure treatments

Generally, mean PVI values of macrophytes in enclosure treatments in both areas A and B were higher than in open areas (*Figure 6.8*). Mean value for PVI in enclosure areas was 0.46±0.34 and in control areas was 0.35±0.24. Overall there were significant differences of PVI values between enclosure and control treatments (F = 9.719, P = 0.002) and also a significant difference between seasons (F = 3.971, P = 0.008). A significant difference of mean PVI values between enclosure and control treatments was also found in area A (F = 7.319, P = 0.008), but not in area B (F = 2.784, P = 0.097).

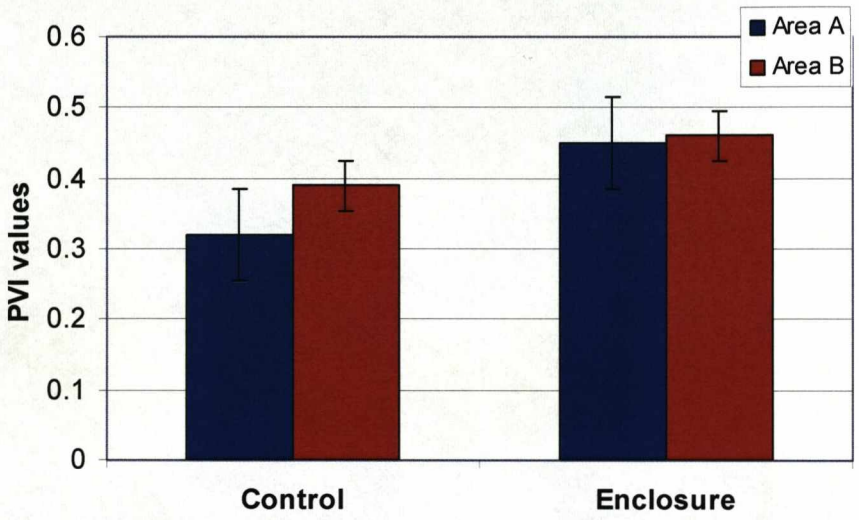


Figure 6.8 Mean of macrophyte PVI values in areas A and B with standard error (n = 2)

Numbers of plant species in enclosure treatments were relatively higher than those found in open areas and statistical analysis indicated a significant difference in numbers of species between enclosure and control treatments (F = 7.935, P = 0.005) (*Figure 6.9*). Dominant species of submerged, emergent and floating plants recorded in both areas A and B included *Alisma plantago-aquatica*, *Oenanthe aquatica*, *Typha latifolia*, *Ranunculus sceleratus*, *Juncus effusus*, *Drepanocladus aduncus*, filamentous algae, *Nitella* sp., *Lemna* spp and *Lemna trisulca*.

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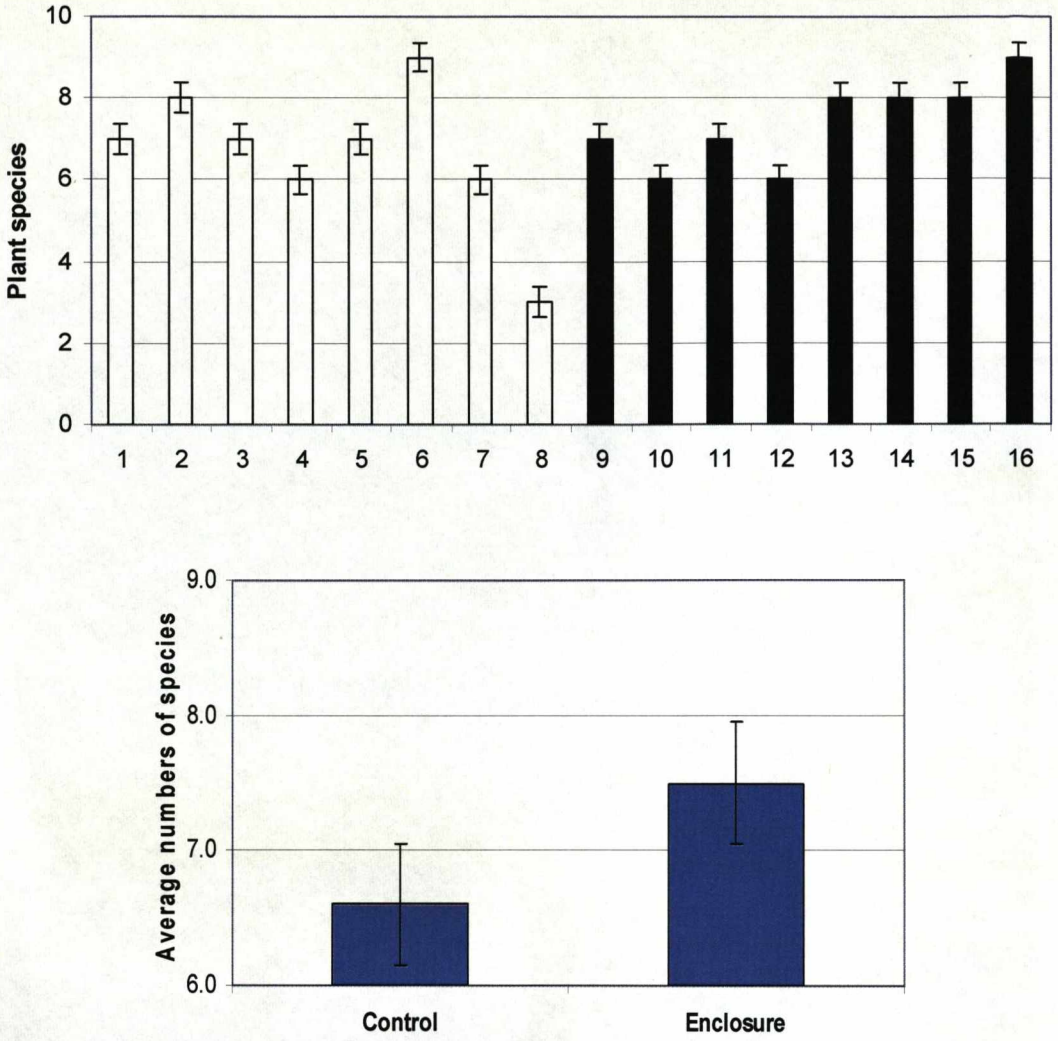


Figure 6.9 Upper graph shows average numbers of plant species in each treatment (white and black columns represent control and enclosure treatments, respectively) and lower graph shows mean number of species of macrophytes in control and enclosure areas with standard error (n = 2).

Differences between mean PVI values in enclosure and control treatments became clear after October 2005 (*Figure 6.4*). Furthermore, mean PVI values between protected and unprotected areas in area A showed a clearer picture than that of area B. One significant influence on the differences of mean PVI values in enclosure and control treatments in area A was likely due to waterfowl.

6.3.2 Waterfowl study

Waterfowl were found in the studied areas throughout the year. Main species of waterfowl in area A were mallard, coot and Canada geese (*Figure 6.10*). Other species such as graylag geese, teal, black headed gulls and shovelers were also present but not in high numbers. Most waterfowl were present in area A rather than in area B. The eleven species of waterfowl in area A included black headed gull (*Larus ridibundus*), Canada goose (*Branta canadensis*), coot (*Fulica atra*), graylag goose (*Anser anser*), little grebe (*Tachybaptus ruficollis*), mallard (*Anas platyrhynchos*), moorhen (*Gallinula chloropus*), mute swan (*Cygnus olor*), shoveler (*Anas clypeata*), teal (*Anas crecca*), tufted duck (*Aythya fuligula*)

In area B, water birds were recorded once in November 2005 (2 mallard), once in March 2006 (4 mallard), and three times in 2007 (two Canada geese in January, two mallard in April and one coot and one mallard in November).

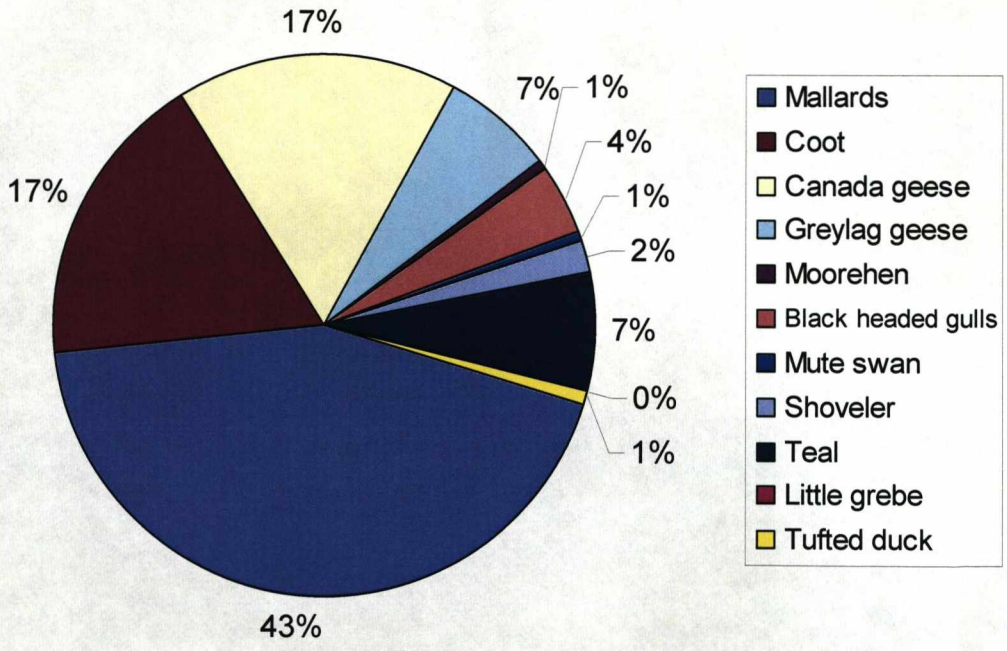


Figure 6.10 Percentage of waterfowl species present in area A

Figure 6.11 shows seasonal changes of the waterfowl population in areas A and B. Numbers of birds tended to decrease in summer and increase in winter. In summer 2005, densities of waterfowl were relatively low compared with other years and the lowest density of waterfowl recorded was 6 individuals per hectare. After September 2005 numbers increased and remained high until spring and then started to decline. In 2006, waterfowl were present in the highest numbers and the highest density of water birds recorded was 110 individuals per hectare in December. In spring 2007, bird populations gradually declined and remained low in summer, but from August onwards the number of aquatic birds started increasing again. Correlation coefficients calculated between numbers of birds and mean PVI values showed no significant correlation in either control (r = -0.007, P = 0.975) or enclosure treatments (r = -0.138, P = 0.552).

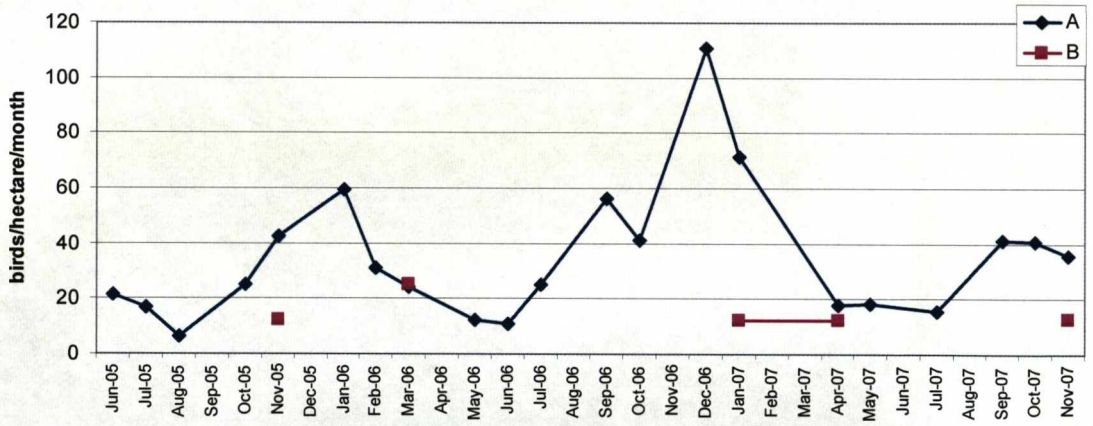


Figure 6.11 Variations of waterfowl densities in areas A and B

6.4 Discussion

Mean PVI values for macrophytes in enclosures and open areas were not different at the beginning of this experiment in summer 2005 when few species of aquatic birds were found and a small number of underwater plants was recorded (*Figure 6.4*). After the pool dried out in summer 2005, the whole area of the pool became overgrown and several species of grasses gradually spread over both enclosure and control treatments. Vegetation in enclosure treatments appeared to grow quicker and better than that found in open areas because they were safely protected from disturbance of not only water birds but also other animals such as rabbits and dogs (*Figure 6.12*). In contrast, macrophytes in open areas were grazed and development of them was poorer than those in enclosure. Our results were similar to Jupe and Spence (1977), Lauridsen et al. (1993) and Sondergaard et al. (1996) who stated that plant height in protected areas was greater than that in unprotected areas. A recent study also showed that biomass of *Ranunculus penicillatus pseudofluitans* was lower in grazed areas (O'Hare et al. 2007).

Figure 6.12 Foot prints of birds and other animals such as rabbits and dogs found around the enclosure treatments

Seasonal changes appeared to influence growth of macrophytes in both enclosure and control treatments since vegetation in the studied areas showed similar trends. In winter 2005-6, PVI values of macrophytes in both enclosure and control treatments were low but in spring PVI values started to increase and remained high until summer (*Figure 6.4*). This was probably because in winter, unfavourable environmental conditions such as short duration of light, low light intensity and low temperature may limit growth of vegetation. In contrast, during the plant growing season in summer, better weather conditions promoted growth of macrophytes and allowed them to grow. Marklund et al. (2002) stated that in a field study, vegetation biomass was relatively low, probably because of the cold and this was consistent with Noordhuis et al. (2002) reporting that the seasonal duration of *Chara* was short, with most of the above-ground biomass disappearing before winter.

However, differences in PVI values in enclosure and control treatments in 2005 were high and likely due to grazing effects of herbivorous waterfowl. Especially during the spring and winter when migrating waterfowl were present, a number of significant differences between plant growth in enclosure and control areas became clear (*Figures 6.4, 6.13*). In October 2005, the population of aquatic birds, mainly mallard and teal started to rise and remained high throughout winter and they appeared to be the major species, grazing on macrophytes in control treatments and in the pool. Aquatic birds such as mallard and teal feed primarily on macrophytes (Nummi, 1993; Guillemain et al. 2000c; Madsen, 1988; Arzel and Elmberg, 2004). Also, grazing pressure appeared to be significant during the day when birds were more active. During the night birds gathered together and rested. This was in agreement with O'Hare et al. (2007) stating that feeding activity of mute swan on keystone macrophytes on the river at dawn and dusk was much lower than during daylight.

Many studies conducted on waterfowl herbivory on submerged macrophytes have suggested that the most obvious impact occurs in autumn and winter when large numbers of migrating waterfowl congregate (Kiorboe, 1980; Sondergaard et al. 1996; Mitchell and Perrow, 1998). In contrast, due to low numbers of aquatic birds in summer, grazing pressure on macrophytes in control treatments appeared to be relatively low compared with that in winter and this was consistent with Mitchell et al. (1994) and Sondergaard et al. (1996) who considered that waterfowl herbivory in

summer was low. Lauridsen et al. (1993), (2002) and Sondergaard et al. (1996) also stated that typically, small reductions of macrophytes coincided with relatively low waterfowl densities. However, other studies have showed that effects of herbivory by waterfowl on macrophytes was apparent from summer onwards when macrophytes reached a maximum standing crop or began to grow (Lauridsen et al. 1993; Moss et al. 1996).

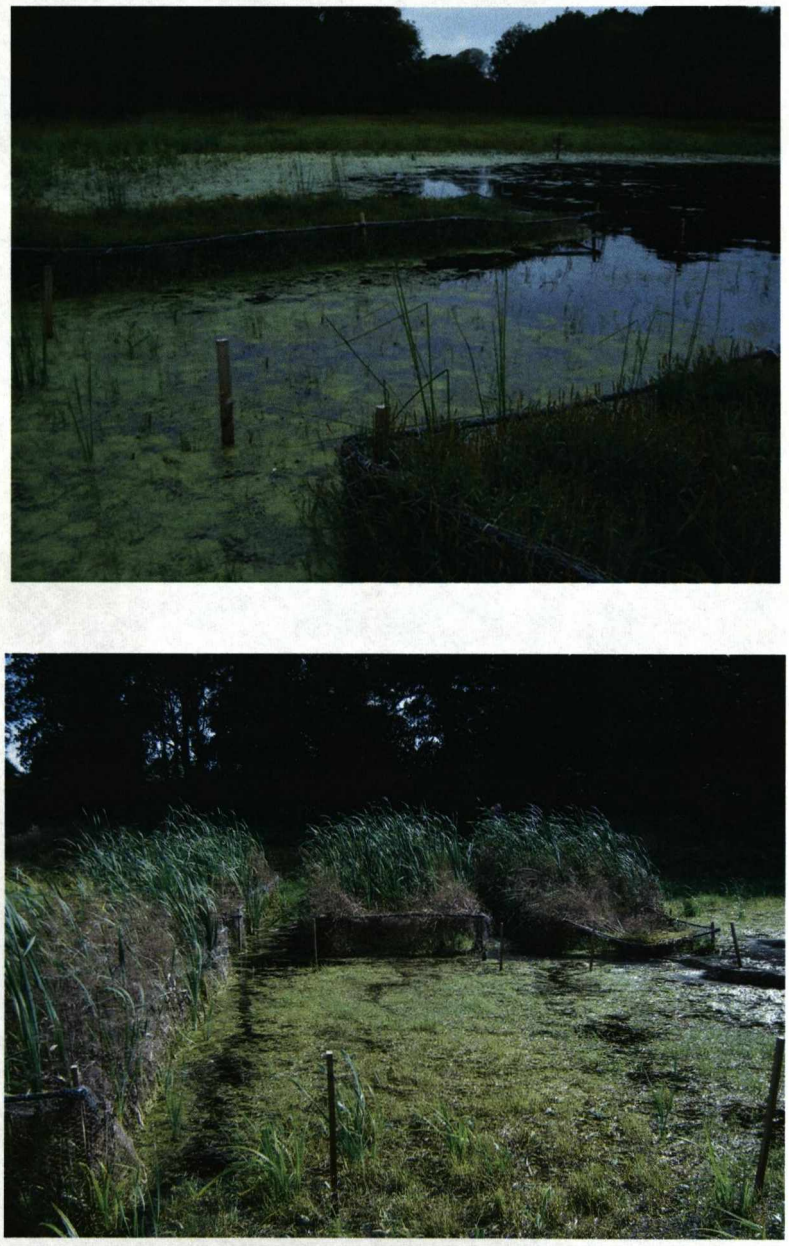


Figure 6.13 Differences of the growth of vegetation between enclosure and control treatments in areas A (above) and B (below) in September 2006

In control treatments, macrophytes were more susceptible to damage because birds could access the plants compared with those in enclosure treatments where macrophytes were safely protected. Therefore, accessibility is believed to be an important factor, influencing food usage of birds and this was in agreement with Euliss and Harris, (1987). An experiment carried out by Noordhuis et al. (2002) confirmed that biomass of *Chara* outside exclosures decreased compared with inside, suggesting a major impact of herbivory on *Chara* biomass. In addition, availability of food sources may attract waterfowl to feed on them. This was seen by observation in 2006 when macrophytes were abundant across the whole lake and populations of waterfowl were highest in this year. A study of Noordhuis et al. (2002) was in agreement with this study as they found that the increase in bird numbers in Lake Veluwemeer, the Netherlands since 1990 was clearly related to the return and increase of *Chara* as *Chara* is the most abundant food resource available to water birds.

Emergent plants in particular were more damaged by birds compared with submerged macrophytes in control plots. Submerged plants were less affected by herbivory and were suppressed when emergent plants were protected and grew well (*Figure 6.5*). This confirmed the fact that birds chose to consume resources easily available and access first. This was in consistent with Schmieder et al. (2006) who reported that waterfowl heavily consumed Charophytes at shallow levels from 0.5 m down to a water depth of 2 m rather than expended more energy on deeper dives at the depth of 4 m as long as enough food resources are easily available in shallow areas. Marchetti and Rivas (2001) also stated that generally submerged plants are not extensively grazed, but the emergents are often attacked by birds.

Statistical analysis demonstrated significant differences of mean PVI values in enclosure and control areas only in area A where most birds were present and this suggested that densities of waterfowl are crucial. Higher densities of birds appeared to cause more damage to aquatic vegetation than lower densities of birds. This was in agreement with Wass and Mitchell (1998) and Marklund et al. (2002) that a reduction of submerged vegetation as a result of waterfowl grazing is related to waterfowl densities. Although no significant differences of mean PVI values between enclosure and control treatments were found in area B, it was noticeable that PVI values of macrophytes in enclosures were still greater than in control treatments. This suggested

that waterfowl did not feed in one particular area but in fact they moved around and fed across the pool where food sources were available. The lack of correlation between bird numbers, recorded at one particular time of day, and PVI does not necessarily deny these conclusions. Most bird activity is at quiet times of day and although the general relationship between activities in areas A and B is likely valid, the attempt at detailed correlation was probably foiled by more complex bird activity than revealed by a single count close to mid-day.

How do birds damage vegetation? Figure 6.14 shows images of aquatic plants destroyed by water birds. Several species of macrophytes such as *Ranunculus sceleratus*, *Oenanthe aquatica* and *Alopercurus aequalis* have been found uprooted and only a portion eaten. Coot especially are known to pull up macrophytes, consume vegetation parts and destroy plant beds (Twilley et al. 1985; Lauridsen et al. 1993). After being damaged, these plants were left floating in the water near the shoreline and they may potentially degrade water quality and cause DO depletion.

Waterfowl also ripped off leaves or other parts of vegetation without uprooting as also reported by Jupp and Spence (1977). But in this case, the plants remain alive and can recover or develop new leaves later, when grazing impact is reduced. Besides the direct consumption of macrophytes by waterfowl, there can be suppression by trampling. Mallard suppressed vegetation in this way.

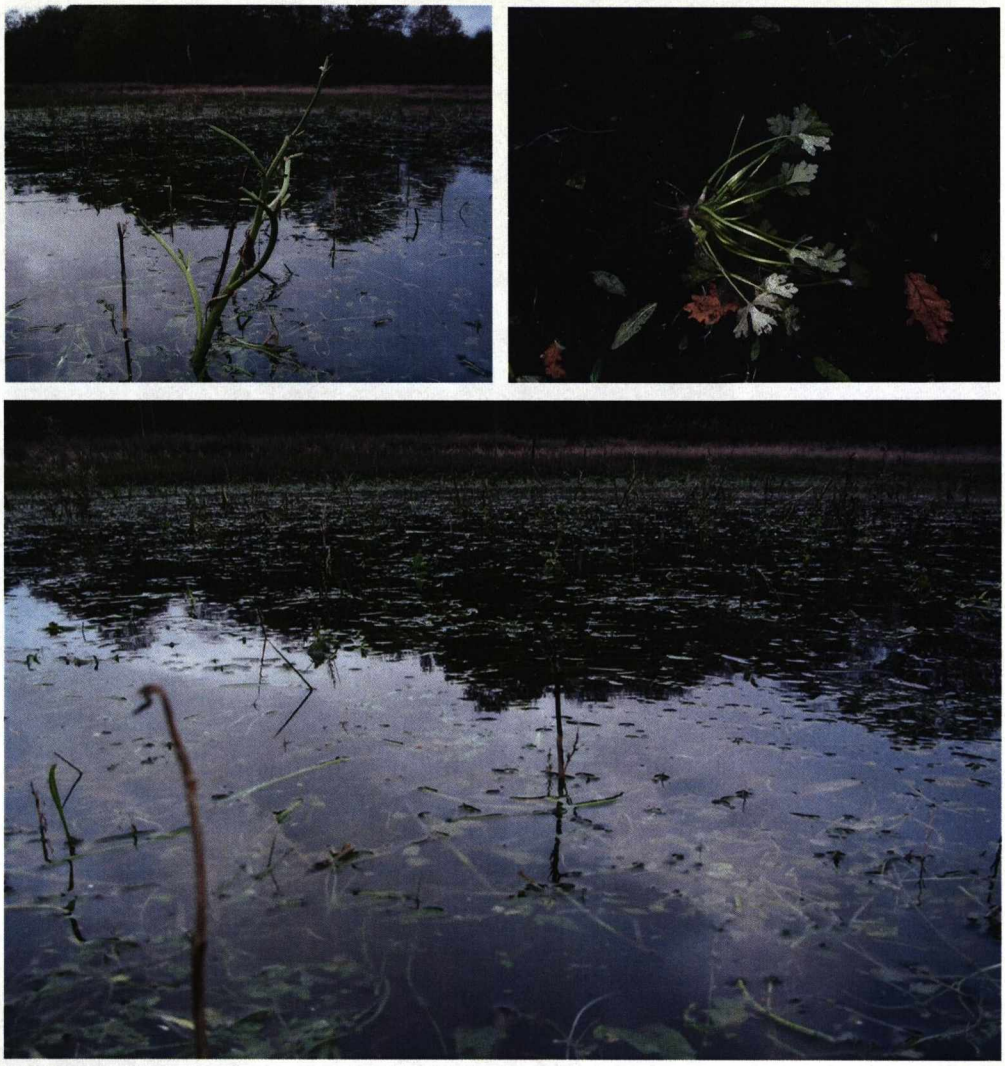


Figure 6.14 Leaves of *Ranunculus scleratus* (above left) have been ripped off and the plant uprooted (above right). General damages of macrophytes across the pool (below) caused by aquatic birds were also observed in October 2005.

Figure 6.15 clearly illustrates the results of trampling on a community of macrophytes including *Juncus effusus* and *Alopecurus aequalis* near the shoreline. Trampling may reduce growth rate and cause physical damage that will lead to slow development of aquatic plant communities.

Figure 6.15 The results of trampling by birds on macrophyte communities in October 2005

Although in summer waterfowl density and grazing pressure in the studied areas was relatively low, big differences between mean PVI of enclosures and control treatments were still found. This suggests that damage and loss of aquatic plants, caused by waterfowl in winter can cause long term impacts on subsequent development of macrophytes in summer. This was in agreement with van Donk et al. (1994) that herbivory by birds in the winter months was partly responsible for the decline of the *Elodea* population and restructuring of the macrophyte community in favour of *Ceratophyllum* in Lake Zwemlust. Also, Sondergaard et al. (1996), Weisner et al. (1997) and Lauridsen et al. (1993) found that herbivorous waterfowl may prevent re-establishment of submersed macrophytes in a shallow lake, thus keeping it in a macrophyte-free, turbid stage.

However, in this case, I did not expect that waterfowl may have a potential to cause an immediate forward switch of a pool from a macrophyte-dominated stage to a turbid stage dominated by phytoplankton. This was because high densities of waterfowl population occurred only in winter and were low in summer. And during the summer growing seasons when there was less grazing pressure, macrophytes can recover. This

was consistent with Hansson et al. (1998), Strand (1999) and Marklund et al. (2002) who reported that the risk that waterfowl grazing would cause lakes to change from a macrophyte-dominated, clear-water phase into a turbid state is low at common waterfowl densities.

Waterfowl not only had a crucial impact on the growth of macrophytes but also affected the number of plant species and composition of vegetation in the lake. More species of macrophytes were recorded in enclosures than those in open areas (*Figure 6.9)*. For example, in area B, *Typha latifolia* and *Alisma plantago-aquatica* were found abundantly in enclosure treatments rather than in control plots (*Figure 6.16*). Absence of such species in control treatments seems to be likely caused by large birds such as swans and geese that they uprooted such species. This finding also corresponded well with other studies stating that bird grazing may have an impact on the macrophyte species composition because species may differ in susceptibility to grazing (Weisner et al. 1997; Noordhuis et al. 2000).

Figure 6.16 Abundance of *Typha latifolia* in enclosure treatments compared with that in control areas (above) and uprooted *T. latifolia* (below) in 2006

In contrast, grazing by birds may be beneficial to macrophyte communities and lakes. For instance, aquatic birds may help remove competitive plants and allow other plants to persist. In shallow lakes in southern Sweden, bird consumption was shown to induce a shift from *Potamogeton pectinatus* to *Chara* spp. and *Myriophyllum spicatum* (Weisner et al. 1997). van Wijk (1989) also found that overexploitation of *Potamogeton pectinatus* would enhance the succession from a *Potamogeton*-dominated to a *Chara*-dominated vegetation. Grazing by birds may also help slow down succession processes of lakes that might become filled or fully dominated by

macrophytes. Beekman et al. (1991) reported that Bewick's swans exploit tubers of *Potamogeton pectinatus* to a certain threshold density below which exploitation is no longer profitable. Another indirect benefit of waterfowl on macrophyte communities is that while grazing, birds may provide fertilization through their droppings for aquatic plants. Some rarer plants are poor competitors and are favoured by disturbance that maintains bare mud areas and removes competitive perennials. *Luronium natans,* which is one such species, has been sporadically recorded at Brown Moss in recent years. Its irregularity may be due to several factors all contributing to the relative abundance of its competitors, including bird activity, drying out and degree of eutrophication.

During winter 2006 – spring 2007, PVI values in open areas were greater than those in enclosure treatments and this was likely due to the disappearance of terrestrial and amphibious plants (*Alopecurus aequalis*, *Atriplex patula*) that were previously abundant in 2005 and 2006. After summer 2006, most terrestrial and amphibious plants started to decline and die in both control and enclosure plots. In 2007 those plants were absent from the experiment and this resulted in low PVI values of enclosure areas. Also less abundance of submerged plants in enclosures appeared to be related to the presence of dense emergent macrophytes as shading from emergent plants reduced light penetration of the water and this may suppress and limit growth of under water vegetation. Marchetti and Rivas (2001) stated that the submerged plants receive less light than emergent ones because of shading. In contrast, in control areas emergent macrophytes were more damaged by birds and this allowed submerged vegetation to colonize and flourish.

In summary, changes in plant growth were mainly due to season but different growth of macrophytes between enclosure and control treatments was likely due to herbivory by waterfowl. Damages of macrophytes were likely due to density of waterfowl, availability of food sources and accessibility of macrophytes. High density of waterfowl in winter resulted in significant differences between mean PVI of enclosure and control treatments and severe damage of water plants appeared to mostly occur during the day in winter when most grazing migratory birds such as mallard and teal were present. Emergent plants especially were more damaged than submerged

macrophytes. In addition, feeding, uprooting and trampling by waterfowl were considered crucial causes of damage to macrophytes in the pool.

Waterfowl also caused long term impacts on the re-development of the water plant community and a reduction of biodiversity of plants (*Fig. 6.15*). However, the effect of waterfowl herbivory has less potential to shift a macrophyte-dominated stage into a phytoplankton-dominated stage as during the growing season, aquatic plants recover and populations of water birds were low. Beneficially, presence of herbivorous waterfowl in the pool may decrease abundance of some macrophyte species and allow other plants to flourish. Lastly, emergent macrophytes appeared to have a negative impact on abundance of submerged plants in enclosures due to shading effects. Effect of waterfowl on aquatic plant communities is presented below (*Figure 6.15*).

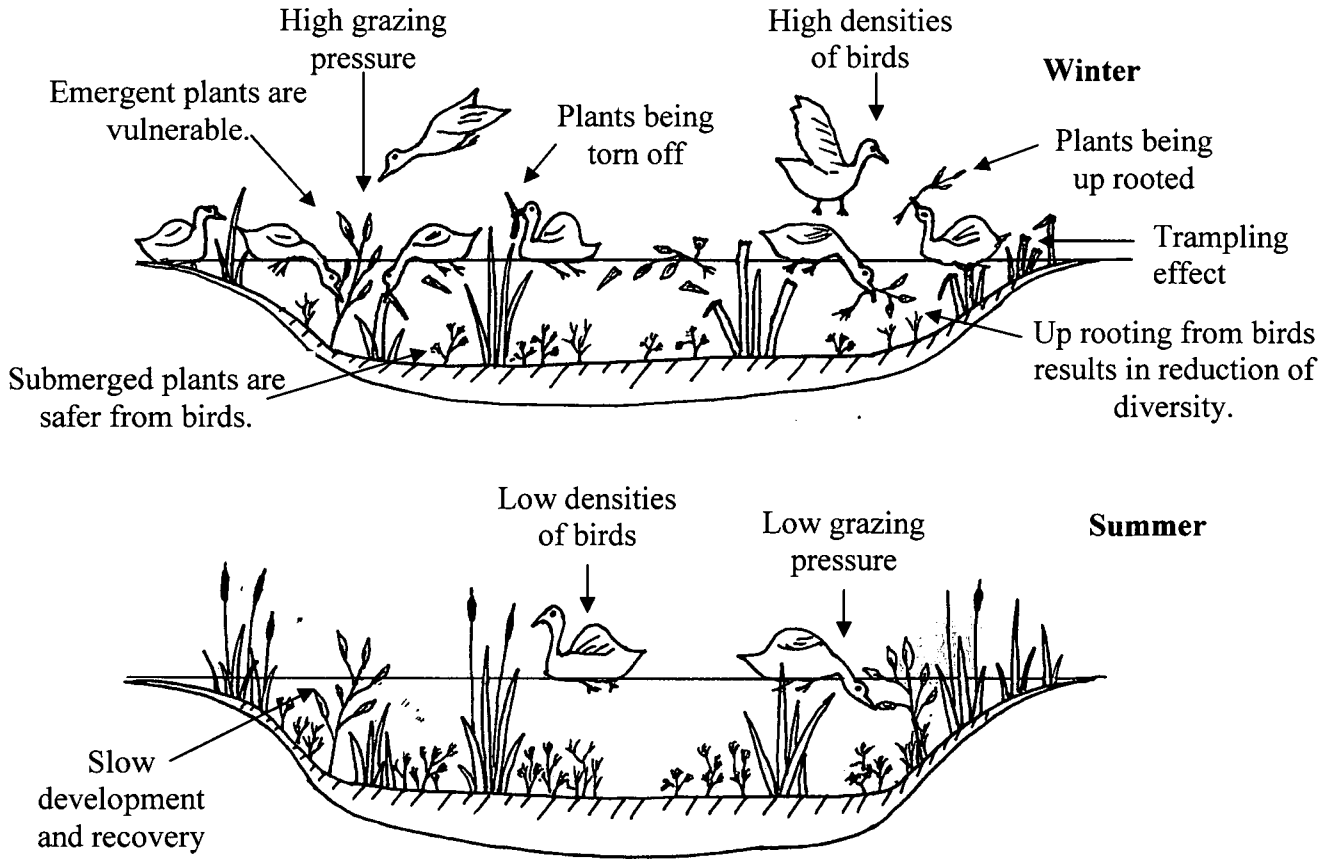


Figure 6.15 Effects of waterfowl on aquatic plant communities

Chapter 7 Paleolimnology of Brown Moss

Chapter 7
Paleolimnology of Brown Moss

7.1 Introduction

Suitable management of aquatic ecosystems and resources requires both short- and long term- and present and past environmental data (Smol, 1992). Present data indicate the current status of lakes whereas historical data can be used to determine background conditions of water bodies. Use of physical, chemical and biological information preserved in sediment profiles is one way of reconstructing past environmental conditions, revealing the ecological history and showing how lakes have developed. Over the last decade, paleolimnology has experienced an infusion of new ideas, approaches and techniques and many of these advances can directly be applied to problems of lake management (Smol, 1992; Hodgson et al. 1998; Reavie, 1998).

Many studies have used paleolimnology to assess environmental impacts and these range from local problems such as eutrophication, soil erosion, and biological response to major changes in environmental variables to more global issues such as climate change (Davis and Berge, 1980; Frey, 1988; Anderson et al. 1990; Fritz, 1990; Autenrieth et al. 1991; Smol, 1992). An example of using paleolimnology to reveal past eutrophication is a study of Lake Arendsee in Germany by Finlay et al. (1998). They found that the lake became significantly more eutrophic between 1900 – 1948 due to human impact and it is therefore conceivable that the present lake reclamation project of calcite additions might be expected to reduce the trophic status of the lake to the level prior to 1900. Similarly, Prat and Daroca (1983) reported that the dominance of *Pediastrum* and *Bosmina* remains in the sediments suggests a fluctuation in water level and eutrophic water in La Brefia reservoir, Spain.

Paleolimnological data can also provide important data on acidification and past lake water pH trends. For example, analysis of subfossil diatom assemblages in sediment cores has been an effective means of inferring past pH conditions (van Dam et al. 1981; Charles, 1984; Cook at el. 1990). Hustedt (1939) divided the diatoms into pH-associated groups ranging from diatom taxa that occur at pH lower than 5.5 to diatoms

that are present only above pH 7. *Fragilaria acidobiontica* (Charles) and *Frustulia rhomboides,* for example, are tolerant species that can occur in acid conditions (Winkler, 1988; Charles et al. 1991). Another study using diatoms as acid indicators was done by Wrinkle (1988) in Duck Pond on the Cape Cod National Seashore, Massachusetts. He found that the pond has been acid for its entire history with a mean reconstructed pH of 5.3 as a result of the effects of acid precipitation. Also, paleolimnological analyses of diatoms and chrysophytes in two hill-top lakes in Norway showed that prior to ca. 1914 both lakes were naturally acid with reconstructed lake-pH values of at least 4.8 – 5.1 (Birks et al. 1990).

Remains of plants (pollen, spores) and animals (cladocerans, Chironomidae) can also be used as indicators of past environmental conditions (Frey, 1960; Battarbee et al. 1999, 2001) but diatoms are of great value. Tibby (2007) suggested that many species of diatoms have clearly defined, and frequently narrow, preferences for particular habitats and water quality conditions. In addition, they are abundant and due to their silica shells they are generally well-preserved and can be used to hind-cast some important aspects of water quality and habitat condition (Lotter, 1989; Reid et al. 1995, 2002). Changes in species composition along dated sediment cores and analysis of species assemblages can lead to substantial advances in the understanding of human impact on aquatic habitats and water quality on timescales ranging from sub-decadal to millennial (Tibby, 2007).

Furthermore, the trophic state histories of lakes can be explored by paleolimnological techniques using organic matter and total phosphorus (Brenner et al. 1999). Sedimentary organic matter can provide records of past and recent environmental changes in local and regional environmental systems (Tenzer et al. 1999; Mayers, 2003). Ariztegui et al. (2001) found that combined sedimentological, bulk and detailed analysis of the organic matter in Lake Albano, Italy indicated high levels of primary productivity. They also revealed that changes in phytoplankton dominance and sharp decreases in productivity are caused by changes in temperature and/or effective rainfall. Tenzer et al. (1999) explored the history of changes in the accumulation of sedimentary organic matter caused by environmental perturbations in the Lake George ecosystem on the Michigan–Ontario border and found that the delivery of organic matter from algal productivity appears to have decreased since the

early part of this century and has been more than replaced by delivery of land-derived organic matter well after the time of forest clear-cutting around 1870.

Paleolimnology may be a useful tool to reconstruct limnological history of pools at Brown Moss. Brown Moss is part of the Meres and Mosses of the N.W. Midlands glacial plain. There are a series of water- or peat-filled hollows in the glacial drift by receding ice sheets which formerly covered the Cheshire/Shropshire Plain around the end of the last Ice Age (Sinker, 1962). As the ice melted away, the intermediate hollows became inundated by meltwater (Reynolds, 1979) and as a result, a series of open water sites, or 'meres', as well as a smaller number of peatland sites, known as 'mosses' where vegetation filled in the shallow basins was created. At an early stage, further fluvian erosions generated by their limited catchment areas would have been insignificant, thus enabling the early postglacial topography to have been preserved as well as it has (Reynolds, 1979). Brown Moss has been conventionally included in the Meres and Mosses.

Brown Moss, in Shropshire, UK, has been a recognisable site for at least 167 years according to the 1841 tithe map of Edgeley Township. The site has a number of designations associated with it which reflect its national and international importance as a wetland area (Edwards, 2007). Because Brown Moss has been recognisable for a long time, the historical uses and management of this site have been well documented as there has been a good range of historical and ecological surveys. Reference to the map in 1841 compared with that in 1997 (*Figure 7.1*) shows significant changes in the outline of water bodies at Brown Moss in the last 167 years (Whild, 2007) as formerly the pools at Brown Moss were larger than at present and there were fewer of them. Currently, the sizes of the pools have become smaller and there are more small water bodies occurring within the site.

Succession and changes in plant communities are also noticeable. Brown Moss was previously registered as Common Land and had been grazed. The area during that period was dominated by grassland or heathland but after the 1960s grazing ceased and as a result secondary woodland has encroached over the heathland (Environment Agency, 2002). The same general trends have also been found in the mere areas such as Hatchmere and Flaxmere near the village of Norley in Cheshire (Lind, 1948, 1949).

In 1953, the area of Brown Moss was notified as a Site of Special Scientific Interest and was owned and managed for public by Shropshire County Council. Since being registered as a SSSI, the site has been visited by a number of people. Fishing for example was a popular activity but in the drought of 1976, the pools dried up and the fish stocks were depleted (Whild, 2007). Other popular activities of visitors at Brown Moss nowadays include bird watching and feeding, and dog walking.

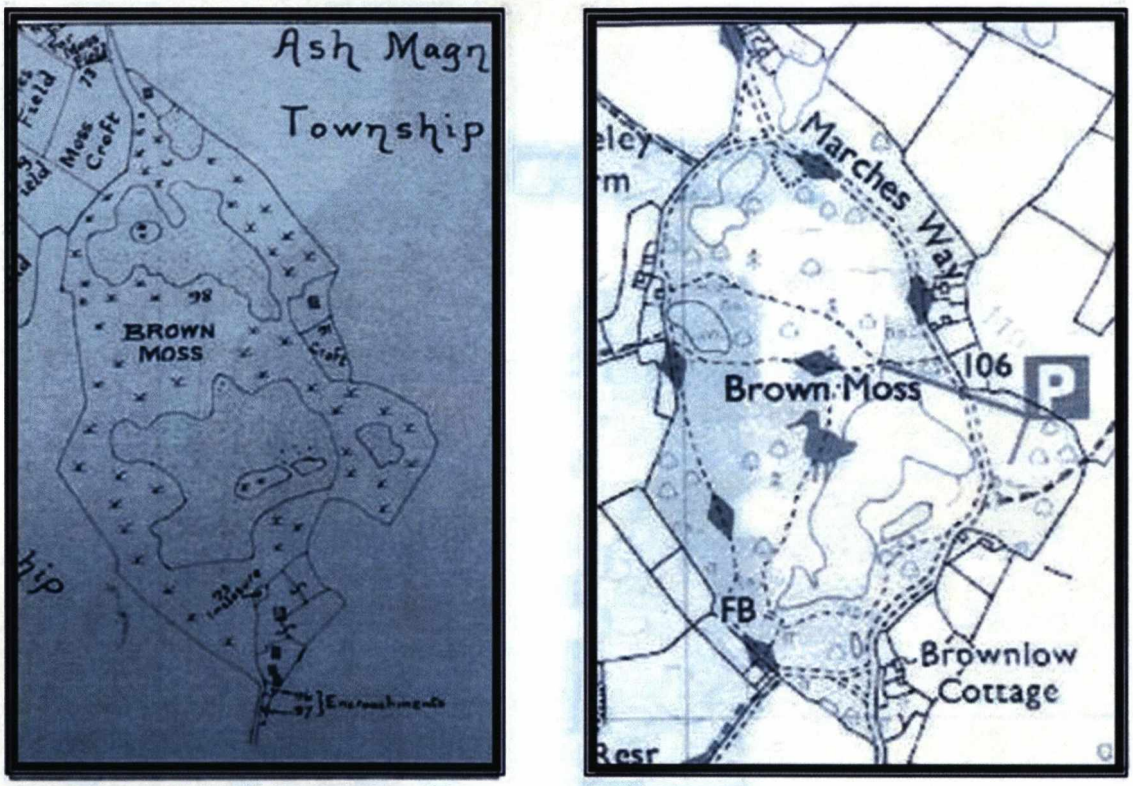


Figure 7.1 Map of Brown Moss in 1841 (left) redrawn by H.D.G. Foxall and the recent Ordnance Survey outline in 1997 (right) (Whild, 2007)

A recent study of Whild (2007) has confirmed a significant change in aquatic vegetation at Brown Moss. She investigated changes in the aquatic vegetation of Brown Moss using historical plant records and Ellenberg values and community analysis. Ellenberg values are widely used to investigate the connection between plant distribution and environmental factors and can be used to monitor environmental change (Hill et al. 2000; Godefroid and Dana, 2007). Each species is allocated a number on a 9- or 12-point ordinal scale according to its distribution with respect to light, temperature, continentality, moisture, reaction, and nitrogen.

Changes in the sum of these numbers for the plant communities may then indicate changes in the relevant environmental factor. The results of Whild (2007) revealed that eutrophic and mesotrophic aquatic plant species at the site have significantly increased. In contrast, significant decreases in oligotrophic species were also detected and this was attributed to an increase in nitrogen loading from agriculture and the atmosphere. These changes were shown to have occurred since the nineteenth century.

Based on findings of Whild (2007), who concentrated on the vegetation as a whole, including the heath and woodlands, I further investigated changes in the aquatic ecosystem of Brown Moss using a preliminary paleolimnological study among the pools. The main purpose of this study was to qualify and quantify ecological history and to determine whether there have been any changes in trophic status of the pools from the past to the present. In addition, comparison of sediment chemistry among the pools was also investigated. Cores were taken from three selected pools at Brown Moss and basic sedimentary chemistry and remains of diatoms were analysed and determined. The results of past environmental conditions of the studied pools may be useful and valuable to the future management of the site.

7.2 Study pools

I selected three shallow pools at Brown Moss to study limnological history. The pools chosen were the floating bog pool (known as pool 9), pools 4 and 6 (*Figures 7.2, 7.3*). The bog pool or so called quaking bog (Whild, 2003) is small and shallow and is almost completely covered by macrophytes. Sinker (1962) described the pool as having a watery moat (*Menyanthes trifoliata*, *Juncus effusus* and *Sphagnum cuspidatum* (Ehrh. ex Hoffm.) surrounding a floating raft of *S. recurvum* in the middle. This pool is one of the most acidic pools at Brown Moss (Whild, 2003). Pool 4 is also shallow and dominated by aquatic plants such as *Juncus* sp., *Polygonum amphibium* and *Carex* sp. Pool 6, the biggest pool at the site, is a natural habitat of aquatic birds. The pool has dried up in some years. Water depth of pool 6 was highest (approx. 60 cm.) among the studied pools.

Figure 7.2 Characteristics of the bog pool (pool 9) and pools 4 and 6 in 2006 (from top to bottom, respectively)

Figure 7.3 Locations of studied pools at Brown Moss

7.3 Methods

7.3.1 Bog and sediment chemistry

A Russian corer was used to collect lake sediments from the bog pool and pool 4. In pool 6, I could not use a Russian corer because water level was high but instead I used plastic tubes (length 50 cm diameter 6.9 cm) with a weighted drop sampler. Core samples in each pool were taken from the deep and shallow areas in summer 2007 and pH of water was also measured in situ. Peat samples in the bog pool were also taken and analysed chemically. After collection, cores from the bog pool and pool 4 were sliced in situ into 1- cm samples on the lake shore. Samples were stored in sealed plastic bags and were brought back to the laboratory at the University of Liverpool. For pool 6, the sediment and water were taken by coring tubes that were pressed into the lake bed. Tubes were then sealed bottom and top with rubber bungs and brought to the laboratory at the University of Liverpool. The water was siphoned off and then a plunger was used to push the sediment slowly from the bottom of the cores until all the sediment had been extruded. The sediments were sliced into 1-cm sections. Chemical analyses of sediment samples were done from the top-lying sediment to 5-cm below the underlying clay layers. All the sediments from each pool were subsampled and processed for further sediment analysis. Methods are given in Chapter 4 section 4.2.1.

7.3.2 Diatom analysis

Diatom samples and slide preparation were based on McGowan (1996) and Battarbee et al. (2001). Diatom study was carried out from the uppermost part of the sediment to the bottom lying sediment layer from deep areas. To prepare diatom samples, I burned sediment samples at 400 ^{o}C in a muffle furnace for 3 hours. Afterwards, ash samples were ground and subsamples of approximately 0.0166 g were taken. Then 10 ml of distilled water was added and samples were shaken well. I pipetted 0.6 ml. of samples onto circular, thickness coverslips and left them overnight until the water had evaporated. The following day, coated coverslips were mounted on glass slides using Naphrax mounting solution. The slides were heated on a hotplate at approximately 130 degree Celsius for about 15 minutes to evaporate the solvent in the resin and then

left them until the Naphrax set hard upon cooling. Diatoms were identified and counted on a Mason microscope, magnification × 200, across entire diametrical transects as distribution is centripetal on drying of a water drop with a prominent meniscus. Measurement of the transect length and width allows calculation of the area counted, whilst transect length gives a measure of the diameter of the cover slip, allowing calculation of the cover slip area. Diatom identifications were based on Wood (1959), Prescott (1980) and (1982).

Calculation of diatom counts (DC, expressed as N g^{-1} of dry weight of sediments) is as follows;

$$DC = V \times \frac{\frac{(N \times A)}{a}}{v \times w}$$

A = area of coverslip

a = area of the transects counted (transect length * field of view width)

N = numbers of diatom on area counted

V = total volume of diatom suspension

v = volume dried on slide

W = dry weight of sediments in preparation

7.3.3 Statistical analyses

I applied Tukey honestly significant difference (HSD) analysis to test for significant differences between pairs of variable means using SPSS 14.0 for Windows. Pearson correlation coefficients (*r*) and regression analysis were also performed to investigate correlation between chemical properties and diatoms throughout the sediment column. Bray-Curtis analysis from PRIMER 6.0 was also used to indicate similarity of variables among the studied pools.

7.4 Results

7.4.1 Peat and sediment chemistry

Peat analysis

The bog pool had floating layers of peat overlying a layer of water and underlying sediment. Analysis of the peat showed that average water content was 95.8±0.3 % (n = 3). A high average content of loss of ignition in peat was also detected at 97.7±0.8 %. Average phosphorus content was 0.3±0.05 mg P dry weight^{-1} and carbonate content was 12±3.7 mg g^{-1}. An example of peat in the studied bog pool is shown below (*Figure 7.4*).

Figure 7.4 Characteristics of surface peat in the studied pool

In the bog pool, the sediment layer under the water lens was approximately 15 cm thick, suggesting low rates of sediment accumulation (*Figure 7.5*). The sediment was black at its surface and gradually became brown and gray towards a basal clay layer. pH value of the water was 3.8, indicating acid conditions. The results of chemical analysis showed that in both deep and shallow areas, contents of all variables in sediments showed a tendency to decrease toward the bottom of the core (*Figure 7.6*). Water content of sediments in deep and shallow areas ranged between 65.5 - 95.1 % and 26.6 – 84 %, respectively. Loss on ignition was high in the uppermost parts of the

core. The lowest and highest percentages of loss on ignition in deep areas were 51.5 and 93.6 %, respectively and in shallow areas were 40.5 and 54.7 %, respectively. Carbonate contents ranged between 11 to 66 mg g^{-1} in deep areas and between 24 to 37 mg g^{-1} in shallow areas. Phosphorus content decreased with increasing depth. The phosphorus content in the top lying sediment was 0.6 mg P dry $weight^{-1}$ in the deep area and was 0.23 mg P dry $weight^{-1}$ in the shallow area. In contrast, the lowest phosphorus contents in deep and shallow areas were 0.2 and 0.003 mg P dry $weight^{-1}$, respectively.

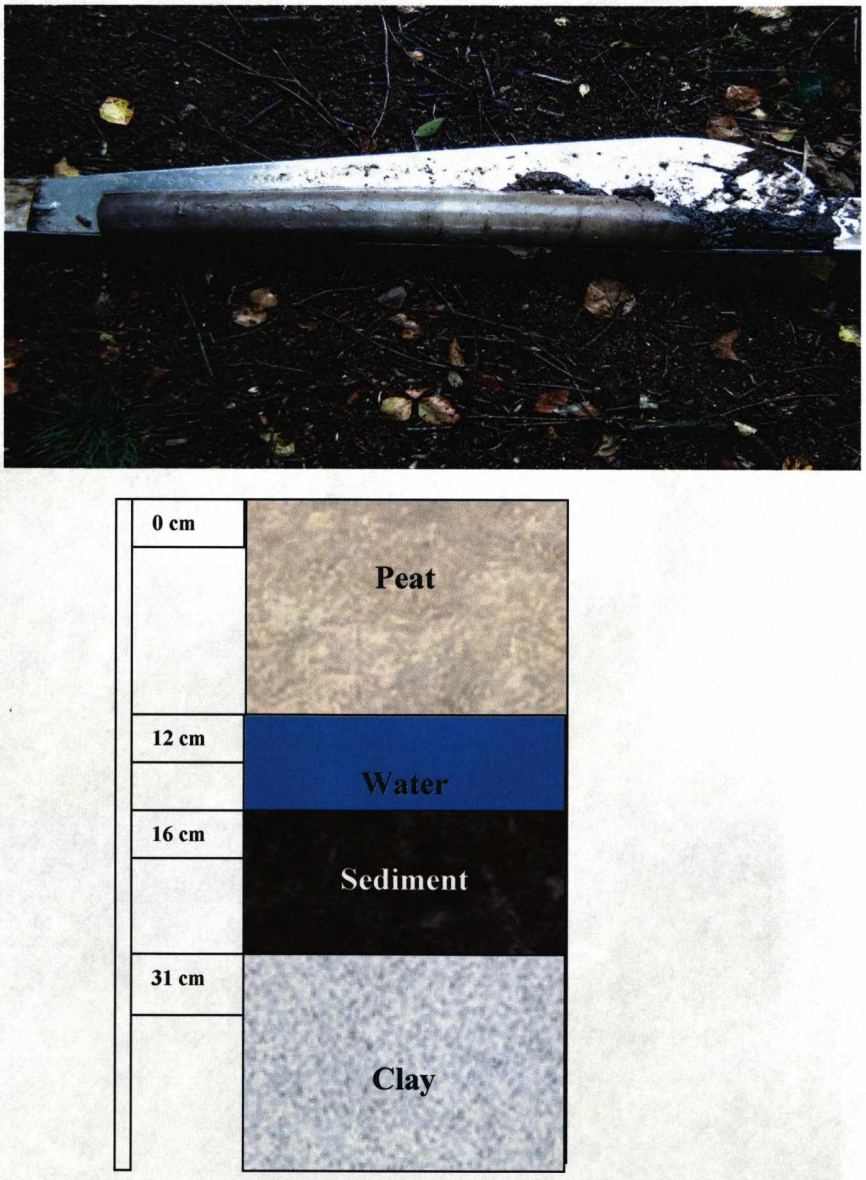


Figure 7.5 Characteristics and approximate depth of sediments in the bog pool (above). The uppermost part of the sediment in the core is to the right. A profile of the bog pool is also presented (below).

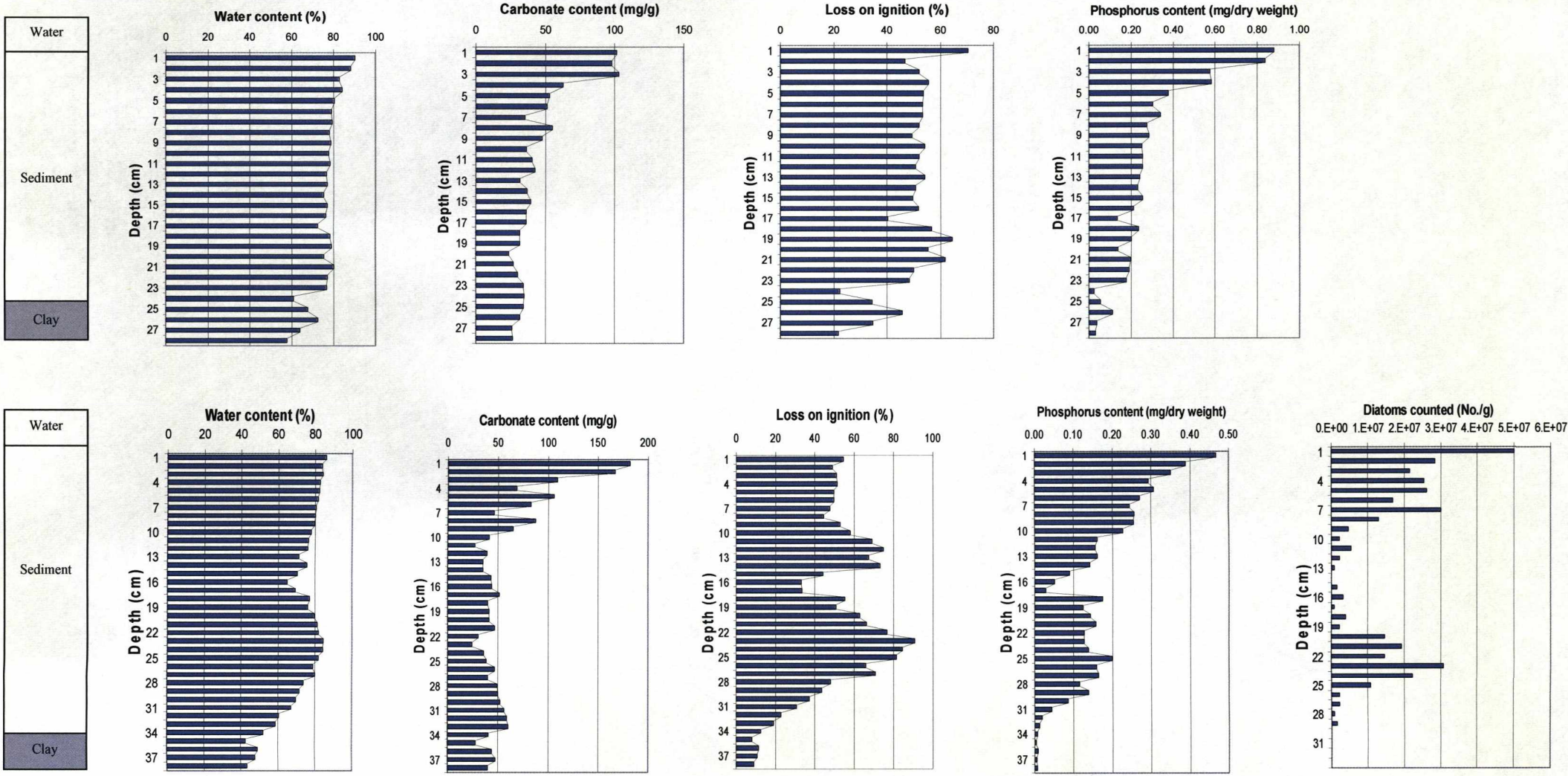
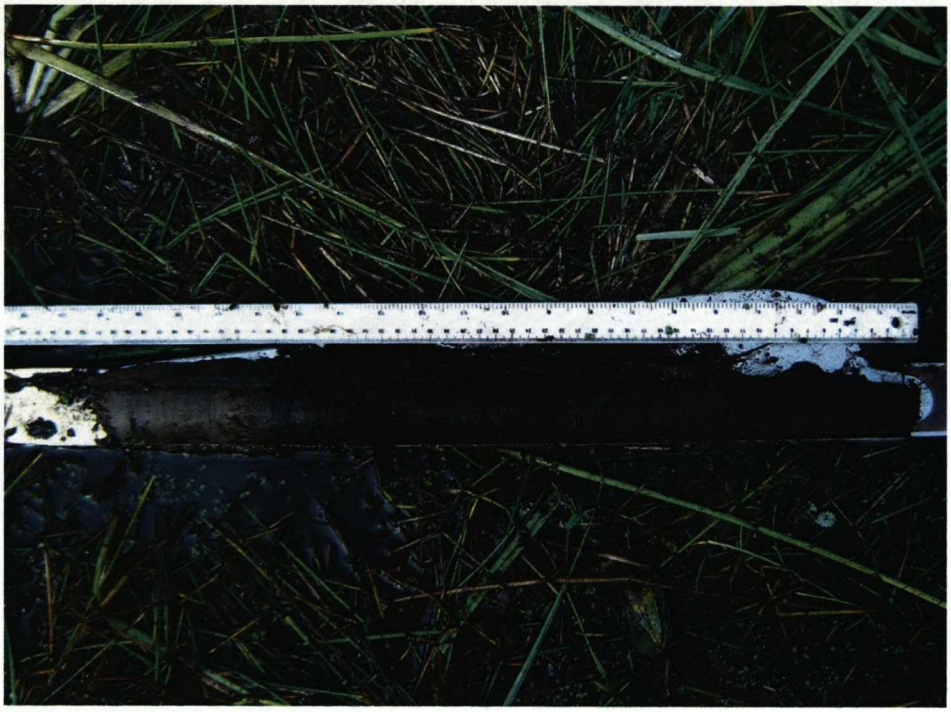


Figure 7.8 Contents of all variables in shallow (above) and deep (below) areas in pool 4

The extent of the sediment layer in pool 4 was about 33 cm and was deeper than in the bog pool (*Figure 7.7*). The sediment layer was black and the underlying clay layer was light brown. pH value of the pool was 6.5. The results of sediment analysis in pool 4 were similar to those in a bog pool in that values of variables were high in top lying sediment and decreased gradually toward the clay layer (*Figure 7.8*). However, there was some disturbance with sharp increases in loss on ignition and phosphorus at around depths of 13 and 23 cm. The increase in loss on ignition was also coincident with contents of phosphorus. Water content from the surface sediment to the lowest layer in deep and shallow areas varied from 43.4 - 85.7 % and from 57.6 - 90.2%, respectively. Loss on ignition in the surface sediment in deep and shallow areas was 54.5 and 70.4 %, respectively and in the lowest layer in deep and shallow areas was 8 and 21.9 %, respectively. From top to lowest lying sediment, carbonate contents in deep and shallow areas ranged between 182 – 60 mg g^{-1} and 101 - 34 mg g^{-1}, respectively. Phosphorus content was highest (0.8 mg P dry $weight^{-1}$) in the surface sediment in shallow areas and in deep areas phosphorus content at the surface sediment was 0.5 mg P dry $weight^{-1}$. At the lowest depth, phosphorus contents in deep and shallow areas were 0.01 and 0.03 mg P dry $weight^{-1}$, respectively.



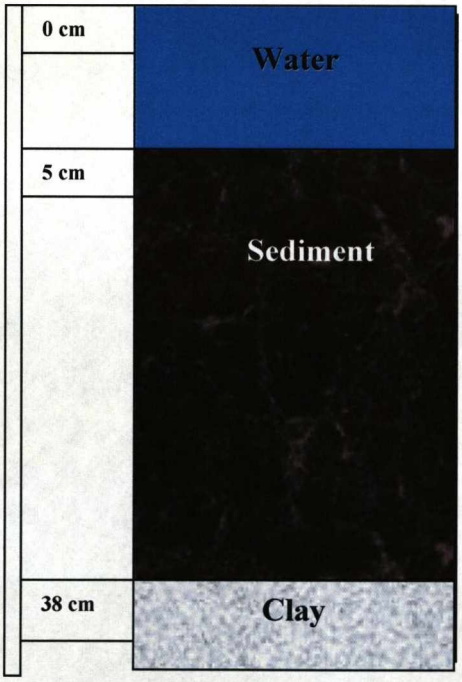


Figure 7.7 Characteristics and approximate depth of sediments in pool 4 (above). The top of the core is towards the right. A profile of the pool 4 is also presented (below).

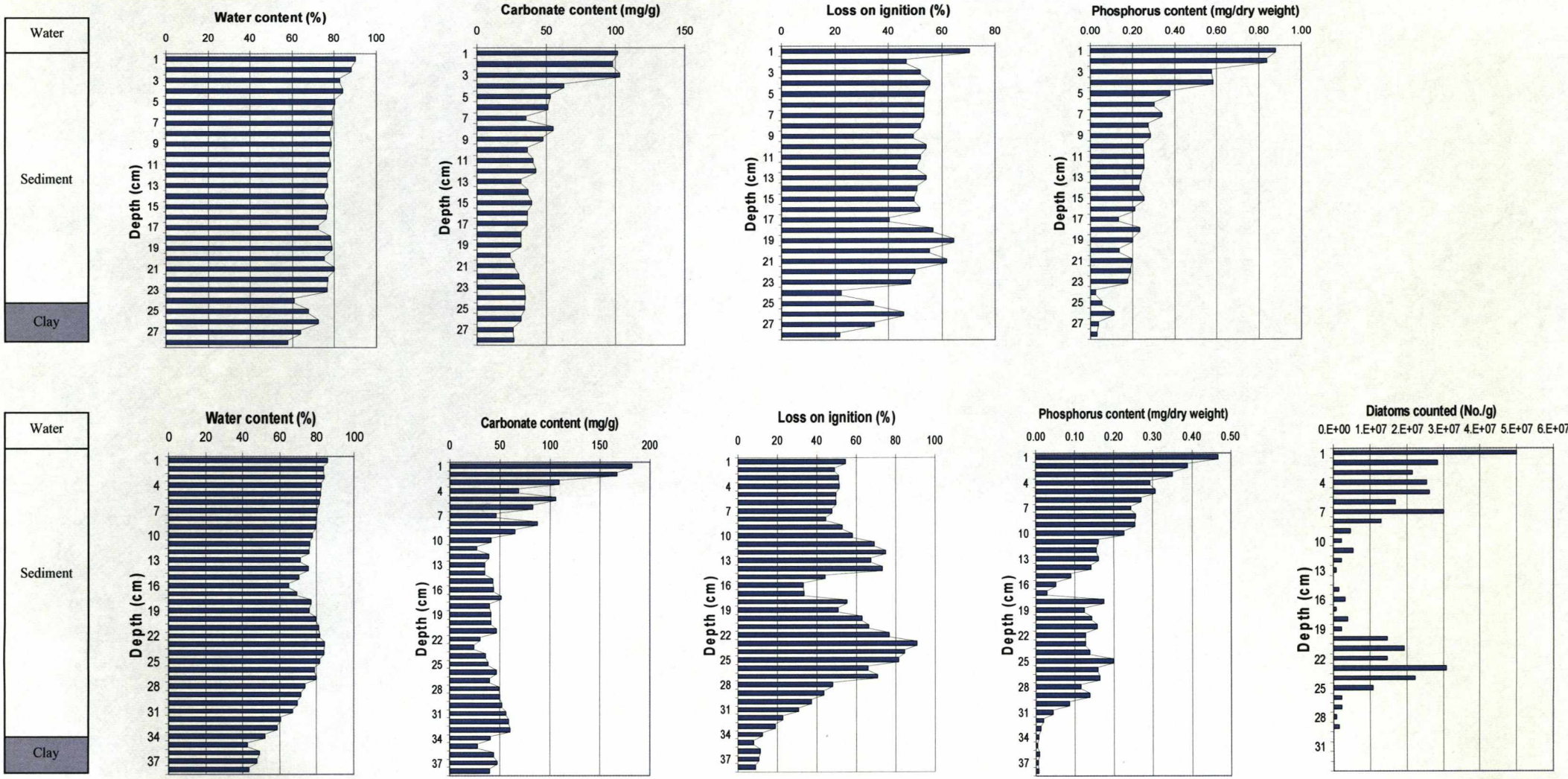


Figure 7.8 Contents of all variables in shallow (above) and deep (below) areas in pool 4

In pool 6, although I was not able to penetrate to the clay layers, the same profile would be expected as in pool 4. The sediment was black and the analysis shows that contents of water, carbonate and loss on ignition were relatively constant but tending to decrease with depth. pH value of the pool was 7.5. Phosphorus content also decreased gradually from the top lying sediment towards the bottom of the core. Water content from the surface sediment to the lowest depth in the deep part of the pool varied from 89 – 75.7% and in shallow areas water content ranged between 86 – 69%, respectively. The results also show that loss on ignition at the surface sediment in deep and shallow areas was 55.9 and 56.1 %, respectively and in the lowest sediment layer in deep and shallow areas was 49.8 and 47.8 %, respectively. Carbonate content in deep areas ranged between 55 – 44 mg g^{-1} from surface sediment to the lowest depth and in the shallow part, carbonate content varied from 72 – 42 mg g^{-1}. Highest phosphorus content was detected in the surface sediment in both deep (0.8 mg P dry $weight^{-1}$) and shallow areas (1.1 mg P dry $weight^{-1}$). At the lowest depth, phosphorus contents in deep and shallow areas were 0.3 and 0.2 mg P dry $weight^{-1}$, respectively.

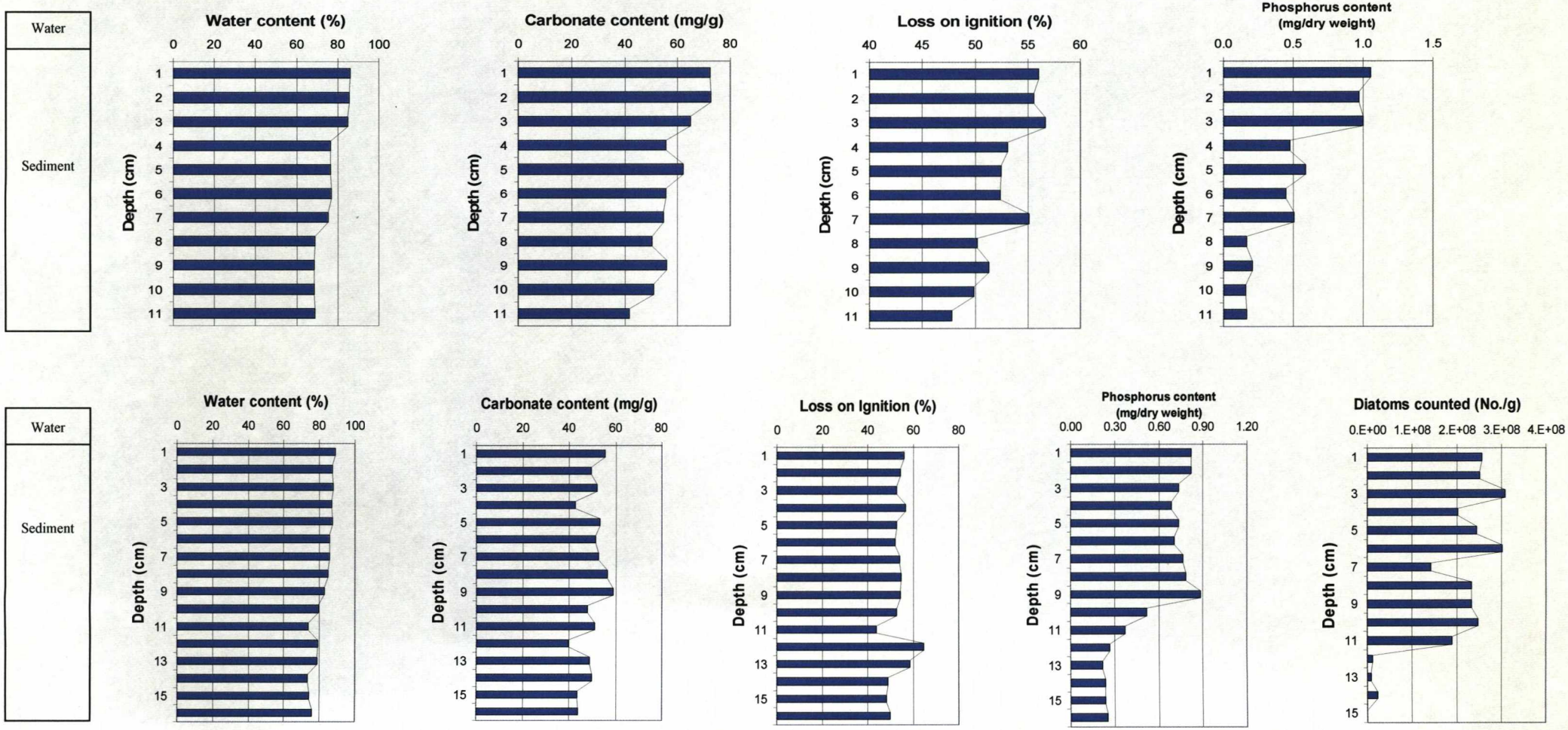


Figure 7.10 Contents of all variables in shallow (above) and deep (below) areas in pool 6

Pearson correlation analysis shows that there were significant positive correlations between phosphorus contents and diatom numbers in pools 4 and 6 in deep areas (($r_{pool\ 4}$ = 0.696, $P_{pool\ 4}$ < 0.001), ($r_{pool\ 6}$ = 0.738, $P_{pool\ 6}$ < 0.001)) but in the bog pool significant negative correlations between phosphorus content and numbers of diatom were found ($r_{bog\ pool}$ = - 0.560, $P_{bog\ pool}$ = 0.03)) (*Figure 7.10*). No significant correlations between loss on ignition and numbers of diatoms were found in all pools (($r_{bog\ pool}$ = 0.384, $P_{bog\ pool}$ = 0.157), ($r_{pool\ 4}$ = 0.181, $P_{pool\ 4}$ = 0.313) and ($r_{pool\ 6}$ = - 0.142, $P_{pool\ 6}$ = 0.615)). Also, there were no correlations between loss on ignition and phosphorus from cores in all pools (($r_{bog\ pool}$ = -0.463, $P_{bog\ pool}$ = 0.082), ($r_{pool\ 4}$ = - 0.150, $P_{pool\ 4}$ = 0.406) and ($r_{pool\ 6}$ = 0.092, $P_{pool\ 6}$ = 0.744)).

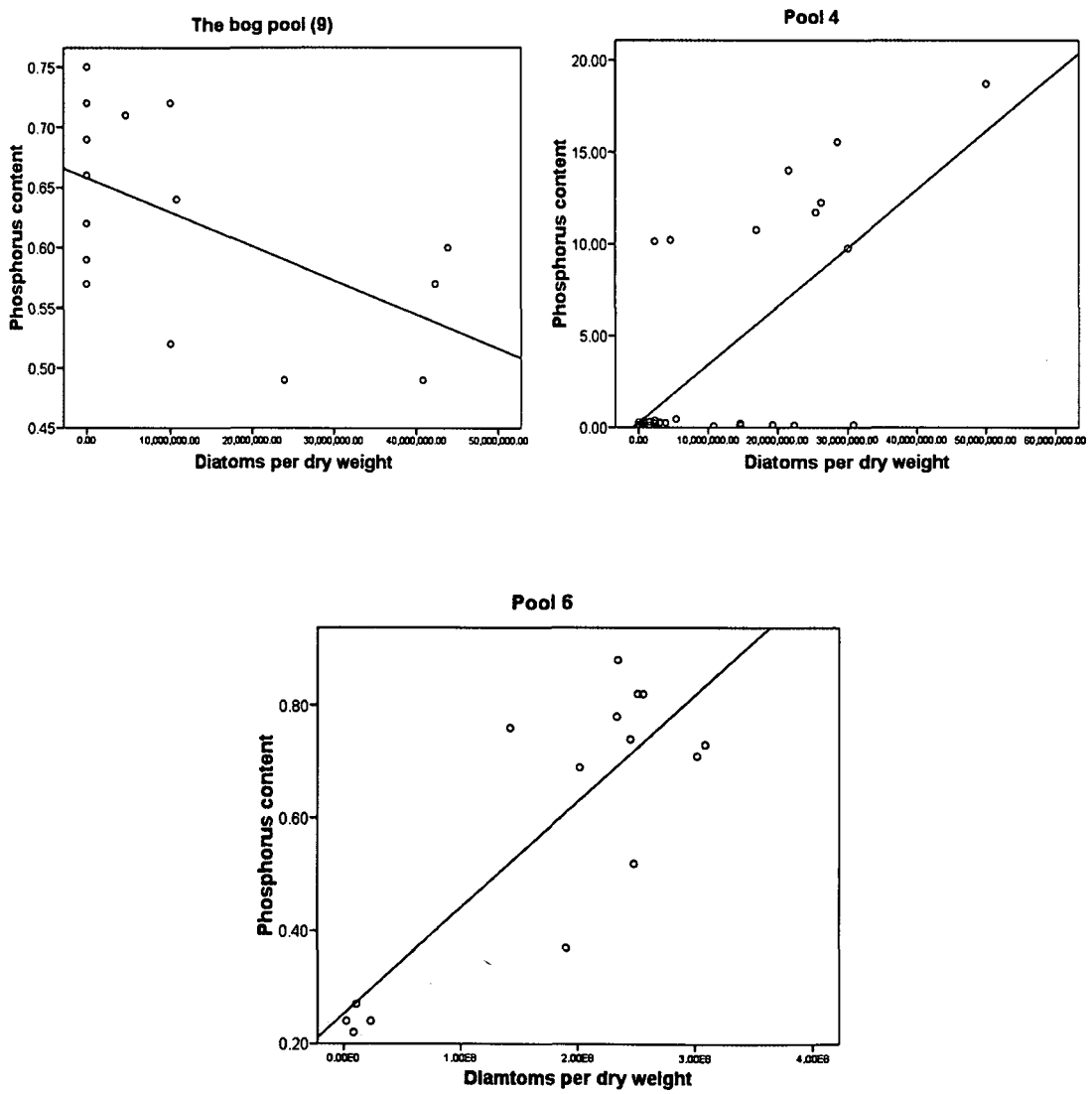


Figure 7.10 Relationships between phosphorus contents and numbers of diatoms in the bog pool, pools 4 and 6 (n = 15)

7.4.2 Comparison of sediment chemistry among the pools

I compared quantities of all variables from deep areas among pools and found that water content of the bog pool was significantly different from that in pool 4 ($P_{bog,4}$ <0.001), but not in pool 6 ($P_{bog,6}$ = 0.477) (*Figure 7.11*). Loss on ignition of the bog pool were significantly different from those in pools 4 ($P_{bog,4}$ <0.001) and 6 ($P_{bog,6}$ = 0.001) but no significant difference of loss on ignition between pools 4 and 6 was found ($P_{4,6}$ = 0.747). Phosphorus content of pool 4 ($P_{4,bog}$ <0.001, $P_{4,6}$ <0.001) was significantly different from the bog pool and pool 6 but no difference of phosphorus content was found between the bog pool and pool 6 ($P_{bog,6}$ = 0.408). Statistical analysis also revealed significant difference of carbonate content between the bog pool and pool 4 ($P_{bog,4}$ = 0.002), but not between the bog pool and pool 6 ($P_{bog,6}$ = 0.086). No significant difference of carbonate content between pools 4 and 6 was found ($P_{4,6}$ = 0.667).

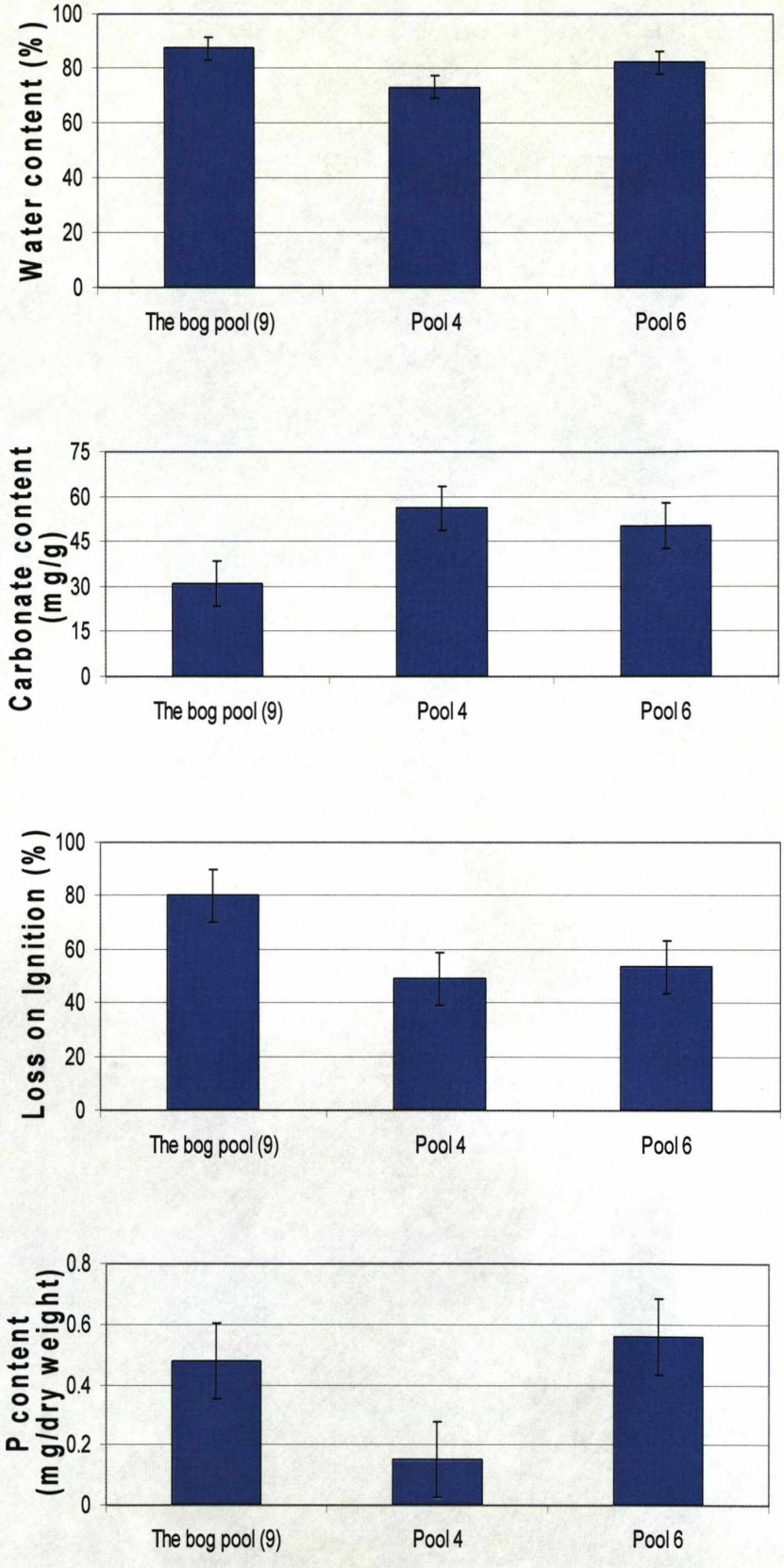


Figure 7.11 Comparative studies of sediment variables in the bog pool, pools 4 and 6 with standard error (n = 15)

Finally, Bray Curtis analysis was applied and showed that variables (water, loss on ignition, carbonate and phosphorus contents) of pools 4 were similar to those in the bog pool (9) with 95.7 percentage of similarity (*Figure 7.12*). Variables measured in the sediment from pool 6 were less similar to those measured in the bog pool and pool 6.

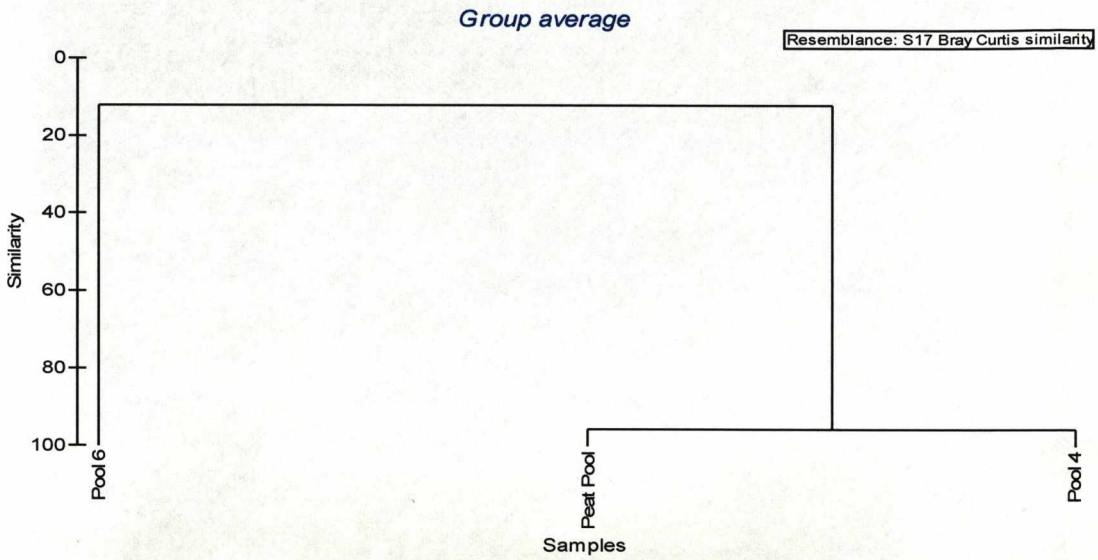


Figure 7.12. Similarity of all variables (water, loss on ignition, carbonate and phosphorus contents) in the bog pool, pools 4 and 6.

7.4.3 Diatom study

Only seven diatom species were found in the bog pool sediments. The main species was *Frustulia rhomboides* and other minor species included *Amphora* spp, *Eunotia indica* (Grunow), *Fragilaria harrisonii* (Grunow), *Mastogloia pseudoparadoxa* (Hust), *Navicula rostellata* (Kutz) and *Pinnularia* sp. In pool 4, there were more species of diatoms than in the bog pool. Thirteen species were found and the dominant species were *Amphora weissflogii* (A. Schmidt) and *Eunotia indica*. In pool 6, ten species of diatoms were found and the dominant species were *Fragilaria harrisonii, Navicula biformis* (Grunow) and *Melosira* spp.

Table 7.1 Species list of sedimentary diatoms found in the bog pool, pools 4 and 6

Species list	Bog	Pool 4	Pool 6
Amphora spp.	√	√	√
Amphora perpusilla (Grunow)		√	
Amphora weissfogii		√	√
Caloneis excentrica (Grunow)			√
Cymbella lanceolata (C. Agardh)			√
Cymbella muellerana (Hust)		√	
Fragilaria harrisonii	√		√
Frustulia rhomboides	√		
Mastogloia pseudoparadoxa	√		
Melosira spp.			√
Navicula bacillum (Ehrenb.)	√	√	
Navicula biformis		√	√
Navicula rostellata	√	√	√
Nitschia spp.		√	
Pinnularia spp.	√	√	
Stauroneis acuta (W. Smith)		√	
Stauroneis schroederi (Hust)		√	√
Surirella biseriata (Bréb. & Godey)		√	
Eunotia indica		√	√

Most diatoms were in the top 10 cm of sediments and numbers of diatoms decreased with increasing depth (*Figures 7.13, 7.14*). Diatoms in the bog pool were absent below about 10 cm. Numbers of diatoms were highest in pool 6 and lowest in pool 4. Average numbers of diatom in the bog pool, pools 4 and 6 were 12,000,000, 11,000,000 and 177,000,000 g^{-1} of dry weight of sediments, respectively. In addition, Figure 14 shows changes in species throughout the cores. Most species of diatoms in all studied pools appeared to increase toward the sediment surface. Especially *Frustulia rhomboides* in the bog pool and *Eunotia indica* in pool 4 increased over the surface 10 cm. Increased numbers of diatoms also appeared to be consistent with increased concentrations of phosphorus.

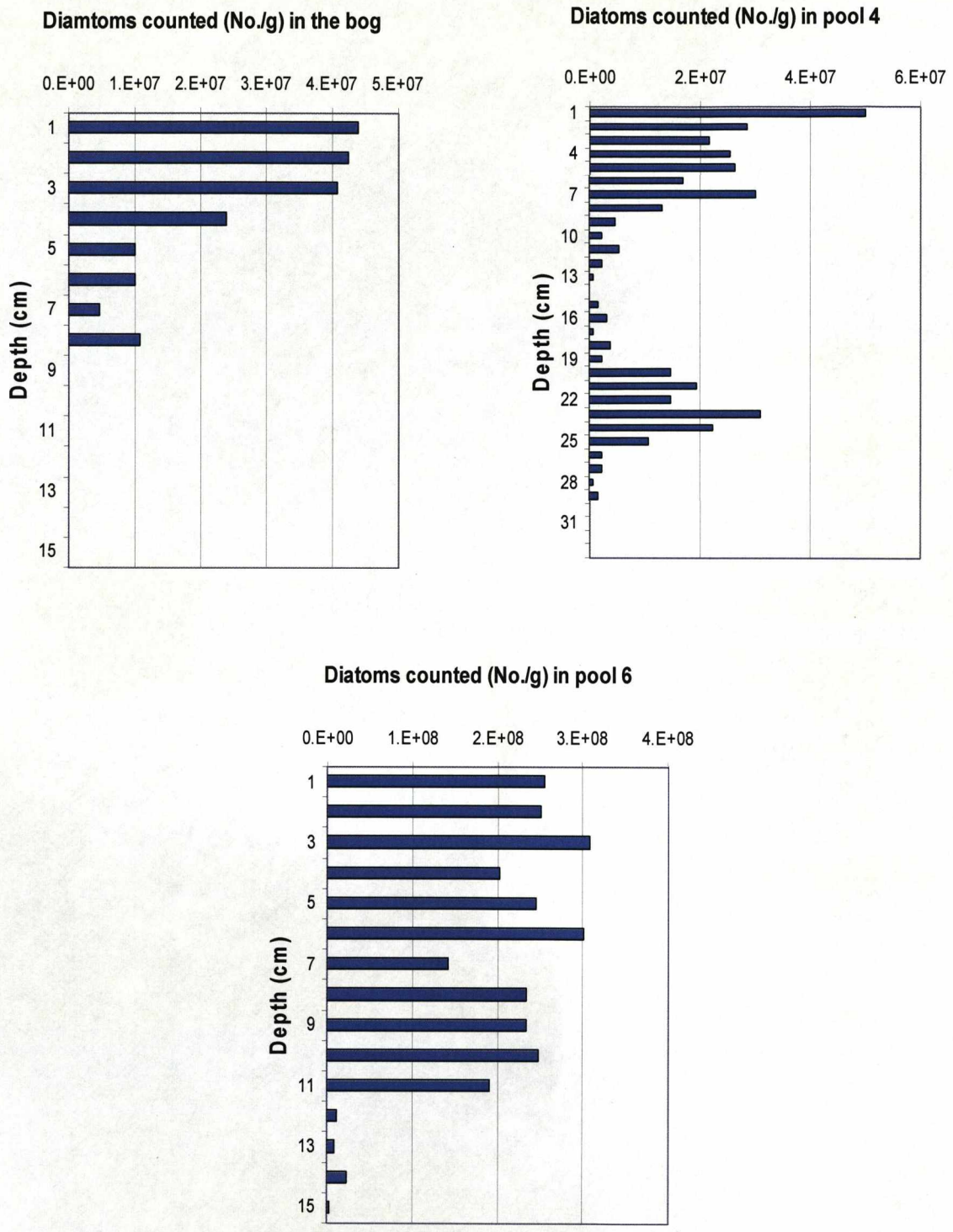


Figure 13. Numbers of diatoms per gram of dry weight of sediments from the bog pool, pools 4 and 6

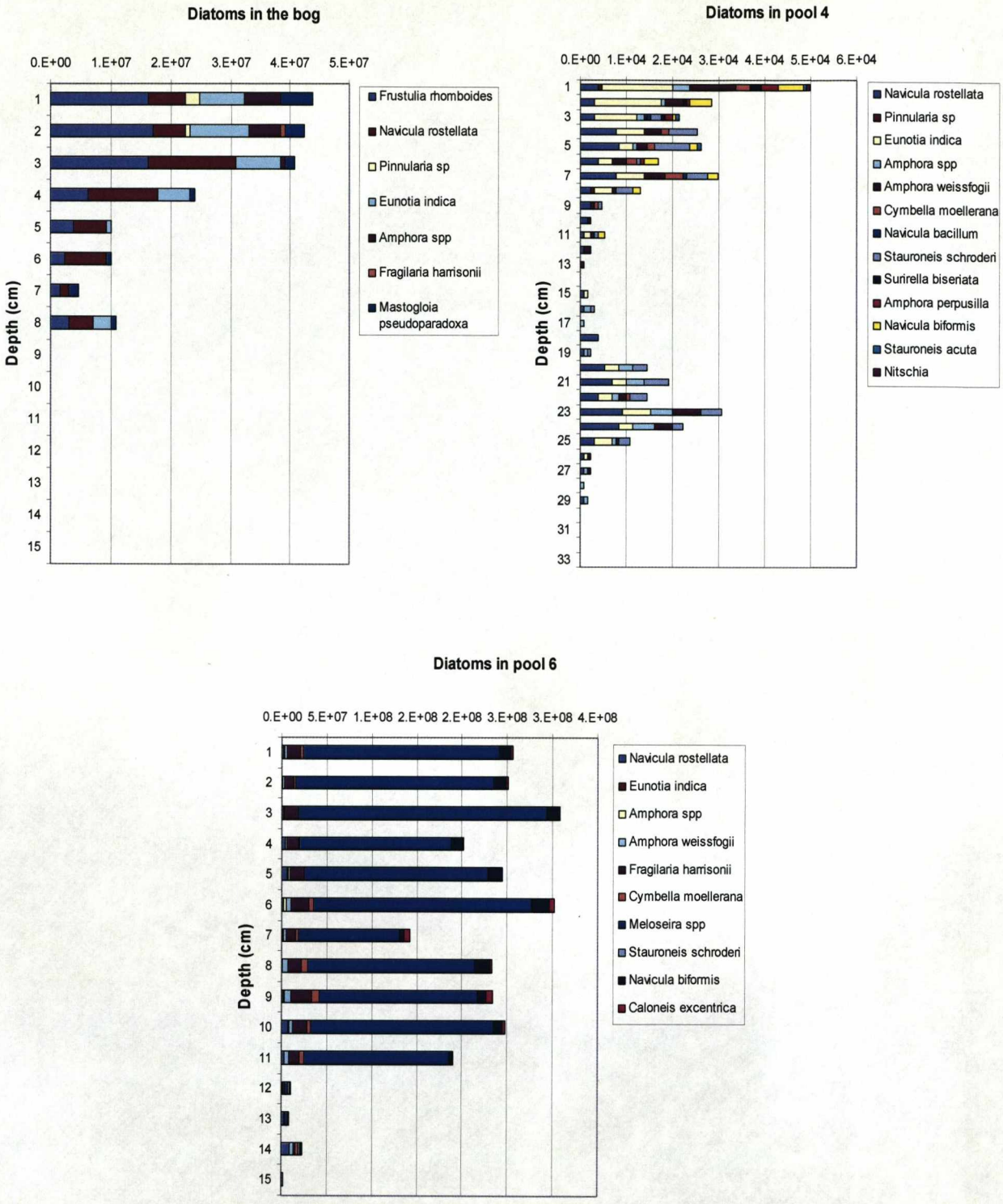


Figure 14. Distribution of diatoms in the studied pools

7.5 Discussion

7.5.1 Limitations and problems

Paleolimnology is a key to reveal limnological history and current trends of pools at Brown Moss. Three different pools were chosen in this study. The bog pool was selected because it represents an undisturbed pool at the site and it is the only bog bog in the area. Pool 4 is a typical small and shallow pool representing other pools around Brown Moss and pool 6 is the main pool. However, limitations of this study were short cores and no dating available. Deep water in pool 6 was also another important limitation in coring samples through to the clay layer. The sediment depths of both the bog pool and pool 4, which have underlying probably glacial clay are very low compared to other lakes. This is likely due to the effects of rapid oxidation of organic matter during the dry phase that prevents sediment accumulation (Collinson et al. 1995) or possibly due to of the past dredging out of sediment or other disturbance. Estimated dates of this study were based on literature reviews of sedimentation rates and depths in other small and shallow lakes (depth below sediment surface around 40 – 50 cm) (Anderson and Odgaard, 1994; McGowan, 1996). Estimated dates of sediments in pool 6 are around 100 to 200 years or back to around the 18^{th} centuries. The sediments in the bog pool may be older however because they have been protected by the mat. It is not really possible to put a date on them. They may conceivably be post glacial as they overlay the clay. The situation in pool 4 is complex as the accumulated sediment layers is thin, yet overlies apparently clay that could be several thousand years old. The sediment may represent an old but disturbed sequence.

7.5.2 Historical and ecological changes in the studied pools

In the bog pool and pool 4, underneath the sediment is apparently the original, smooth glacial clay, implying that the whole area may have been a bigger lake or a wetland. This is consistent with a map of Brown Moss in 1841 showing that there were only two big lakes at the site which formerly could be the same one big lake. Diatoms were scarce in the early processes of pool development after the ice had melted and after the sediment layer began to form. Afterwards, they increased from the bottom of the cores upwards. This may mean that the earliest conditions were probably

unfavourable, disturbed, perhaps through drying out of the pools and wind blow. This is consistent with Moss (1998) explaining that during the late glacial phase, clay and gravel are washed out from under the melting glacier and this may cause a lake to be turbid with suspended clay. Therefore, light can hardly penetrate and photosynthesis is limited and this could cause poor development of phytoplankton. This is often followed by a highly fertile phase in which diatoms increase, based on leaching of abundant nutrients from firstly exposed rock debris.

In the later sediments, organic matter and phosphorus increased together with the diatom count. This may reflect lesser dissolution of recent diatoms or progressive increase in fertility. The similar patterns of phosphorus suggest this as the more likely explanation. A possible explanation might be that as the water cleared, photosynthesis became possible and organic sediments formed as described above for the immediate post glacial period. This explanation would involve loss of all subsequent sediment layers, which seems hardly likely. In case 4 are two phosphorus and diatom peaks, with a trough between them coinciding with sediments of lower water content. This suggested disturbance of some kind.

There were fewer species of diatoms in the sediments in the bog pool than in the other pools possibly due to the acidic conditions where other less tolerant species can not develop or perhaps it may be linked to lower nutrient loading in this pool. The main species of diatom in the bog pool was *Frustulia rhomboids*, which is normally found in bog pools and is an indicator of low pH conditions typical of bog environments. Our result was similar to Walker and Paterson (1986) who stated that the diatom flora of bog pool sediments is extremely limited. *Frustulia rhomboides* is the usual dominant with *Navicula subtilissima* (Cleve) and *Pinnularia* and other taxa usually occurring as minor constituents of diatom flora. Other studies have also found *Frustulia rhomboides* in sediments of acidic lakes (Meriläinen et al. 1982; Birks et al. 1990).

The quantitatively limited diatom flora, with its implications of low nutrient loading, and the development of a bog at this site raises questions of how such extreme habitat differentiation may have occurred within the site as a whole. The bog pool is relatively close to other pools of very different character. One possibility is that for

random reasons this small area remained undisturbed when the rest of Brown Moss was being exploited or polluted by sources of nutrients from effluent or agriculture. The bog pool lies centrally and the rise in diatoms through the short sediment layer suggests some influence of increased nutrients even so. Development of further sediment must have been precluded by closure of the floating peat mat but we have little information as to whether this happened a long time previously or relatively recently or it could be the result of an acidic condition that reduces mineralisation rates of organic matter (Thomann, 1972). The chances are that it was an ancient event and development of the treacherous floating mat may have protected the area from disturbance by humans or cattle that would have been loath to venture upon it when sounder ground was available nearby.

In pool 4, the layer below the sediments is also clay suggesting that the pools at Brown Moss had the same origin. Generally the development of this pool was similar to the bog pool as mentioned earlier. Diatoms were absent from the sediments just above the clay layer suggesting that the pool in the early phase was infertile or turbid or both or that the earliest sediments have been lost. After that, the lake sediments were richer in nutrients and carbonate started to build up, perhaps washed out from surrounding land or deposited as marl by aquatic plants. The build up of diatoms, organic matter and TP was temporary however and then reversed to give a period of low production. It is possible that this period represents the early fertile phase of glacial lakes, but the short length of the sediment core argues against this. Glacial lake cores are usually several metres in length not a mere 37 cm. The later second rise in indicators of increasing production could then be a reflection of more recent eutrophication, or both peaks could be recent phases of eutrophication. It is strange that two peaks were not also recorded in the bog pool, but the absence of the second may reflect closing over of the peat floating mat as discussed above.

The upper peak and also the trend in the cores from pool 6 suggest an increase in productivity possibly from human activities (Macan, 1970; Moss, 1998). From some earlier period Brown Moss land was grazed by domestic stock (Whild, 1996). This would lead to both manuring and higher rates of erosion and accumulation of sediments. There would also have been changes in catchment land use practices after settlement around the site. McGowan (1996) also found the large increases in loss on

ignition at the middle point of the cores of White Mere in Shropshire suggesting that land disturbance occurred. She mentioned that there were two such events causing land disturbance, probably relating to firstly the clearance of the forest and then to a subsequent ploughing of the land. O'Sullivan et al. (1991) also stated that erosion of catchment material, especially topsoil from arable fields, associated with the intensification of agriculture has resulted in an increase in sedimentation within Slapton Ley, a shallow eutrophic lake in SW England. Sedimentation rates vary a great deal from lake to lake but are usually of the order of a few millimetres per year. The likelihood is that the sediment cores from pool 4 represent the last two to three centuries and hence a changing agriculture rather than immediately post-glacial events.

In comparison with the bog pool, pool 4 had a greater variety of diatoms. Walker and Paterson (1986) found that the number of taxa which may occur in weakly acidic lakes is great and taxa which appear to be widely distributed in this environment include the following; *Achnanthes, Cymbella, Eunotia indica, Fragilaria, Navicula* and *Melosira*. This was in agreement with this study in that *Eunotia* were most abundant in pool 4 and the increase in diatoms corresponded well with contents of phosphorus. The local eutrophication of pool 4 with higher numbers of diatoms and protection from disturbance by the quaking surface of the bog pool with lower numbers of diatoms have proven this.

The sediments of pool 6 almost certainly represent recent history of Brown Moss. There was a monotonic increase in organic matter and calcium carbonate because of marl formation or increased erosion and this corresponded well with increased production as indicated by diatoms. The increased phosphorus content was also observed in the surficial sediments and this may be related to increased agricultural activity and impact from the catchment. Froelich (1988) showed that catchment erosion can increase sediment concentrations because phosphorus attaches to a wide variety of mineral surfaces. In pool 6, numbers of diatoms were higher and similar to other high nutrient lakes such as White Mere ($18 \times 10^7\ g^{-1}$), England and Eaglet Lake ($14 \times 10^7\ g^{-1}$), in Canada (McGowan, 1996; Cumming et al. 2006). The variety of diatoms was also greater than in other pools. This is likely due to the fact that pool 6 is more fertile and less acidic than the bog pool. *Melosira* were most abundant in

sediments of pool 6 and is an indicator of eutrophic conditions (Crabtree and Round, 1967).

Overall, there is evidence of some increase in production of all pools. The study of water chemistry (Chapter 2) and nutrient budget (Chapter 3) also confirmed that the pools at Brown Moss had high nutrient concentrations. Concomitantly, the pools have become smaller as they have filled in and as vegetation has encroached at the edges. Filling in is not necessarily accompanied by eutrophication, as evidenced by the bog pool with its sparse diatom content and highly acidic nature, but the two processes have probably been occurring together in pools 4 and 6. Changes in land use and practice and clearance of forest in the catchment area may influence these processes. A marked increase in diatom numbers associated with phosphorus contents in recent uppermost sediments reflected changes in trophic status of pools. The degree of eutrophication of the site has been increasing and greater than in the past and this could be related a result of anthropogenic disturbance of the catchment or natural succession or both (Reynolds, 1979; McGowan, 1996; Anderson and Odgaard, 1994). This finding was also in agreement with Whild (2007) who found that there has been significant change in the Meres in general from sites supporting oligotrophic assemblages to sites that support mostly eutrophic assemblages of higher plants.

Chapter 8 Implications for future conservation and management of Brown Moss

Chapter 8

Implications for future conservation and management of Brown Moss

8.1 An overview

Brown Moss is composed of a series of standing water bodies that are crucial natural habitats for aquatic plants and animals and has had high conservation value though it is now much reduced. Overall, most pools at Brown Moss were hypereutrophic with exceptional concentrations of nitrogen and phosphorus. Nutrient loading from different potential sources affected water chemistry in the main pool 6 on different occasions. Nitrogen and phosphorus concentrations in pool 6 were maintained in winter by external loading mostly from birds, shallow ground and soil water and rain. During summer, external load was relatively small and was less important in nutrient input into pool 6. In contrast, internal load became crucial through sediment release.

Most nitrogen in the main pool came from surface runoff and ground or soil water. One potential nitrogen source was agriculture as fertilizers applied can be transported by surface run off and shallow ground water to the pool. Volatile nitrogen from agricultural areas and vehicle exhausts was also likely to result in high concentrations of N in rain water, which when it fell could increase N concentrations. For phosphorus, sediments appeared to be the main source. Sediment composition revealed that most phosphorus occurred in top lying layers and was released to the water column mostly in summer. The mechanisms involved in release of nutrients included physical, chemical and biological processes. Black sediments in pool 6 indicated an anaerobic condition and under such conditions phosphate sorbed with iron (III) is released. Bioturbation caused by benthic animals such as chironomid larvae and decomposition of organic matter by microorganisms were also likely to cause nutrient release. In contrast, in small pools connected to pool 6, sediments acted as a sink of nutrients with calcium carbonate ($CaCO_3$) in the sediment precipitating some phosphorus, but with the sediments more aerobic allowing greater retention of phosphorus by iron compounds. Phosphorus in the sediments will be mainly derived from external sources, the existence of which is a pre-requisite for generating on internal loading during summer.

As Brown Moss is a part of a RAMSAR site and many pools around the area are natural habitats of migrating and residential aquatic birds, there is concern about waterfowl populations that may cause environmental problems to the pools. The observations and experiment on impact of roles of waterfowl on water chemistry and aquatic vegetation showed that most waterfowl were present on the pool in winter and numbers declined towards summer. This pattern occurred in both years of study. However, numbers and variety of birds differed from year to year. Canada geese were dominant in winter 2004-5 but in winter 2005-6, mallard and teal were most abundant. When birds, especially Canada geese gathered in large numbers in pool 6, quantities of nutrients, especially ammonium nitrogen, immediately increased. Organic matter from aquatic birds also was introduced and this was likely to cause problems later in summer when it decomposed. It was therefore suggested that the major external load also came from waterfowl.

As birds feed on aquatic vegetation, the large flock of mallard and teal in pool 6 was also likely to affect communities of macrophytes. The enclosure experiment showed that aquatic plants in enclosure treatments were safely protected from being damaged by birds. In contrast, aquatic plants exposed to birds were more likely to be eaten and destroyed. More grazing pressure appeared to occur during the day when birds were present everywhere in the pool compared with that at night when birds aggregated in a group on a shoreline. Birds do not affect only the growth and well being of vegetation, they also caused loss of biodiversity of aquatic plants as indicated by more species of plants present in enclosures against bird access. Disappearance of plants could be the result of them being uprooted. However, problems of birds damaging macrophytes are reduced when numbers of birds were low in summer and this allowed plants to recover.

Some pools at Brown Moss were plant dominated with clear water while others were very turbid with phytoplankton dominance (*Figure 8.1*). Regional and local factors appeared to be involved in distinctive limnological features of those pools and these factors included climate, land use, presence and absence of macrophytes, zooplankton and fish.

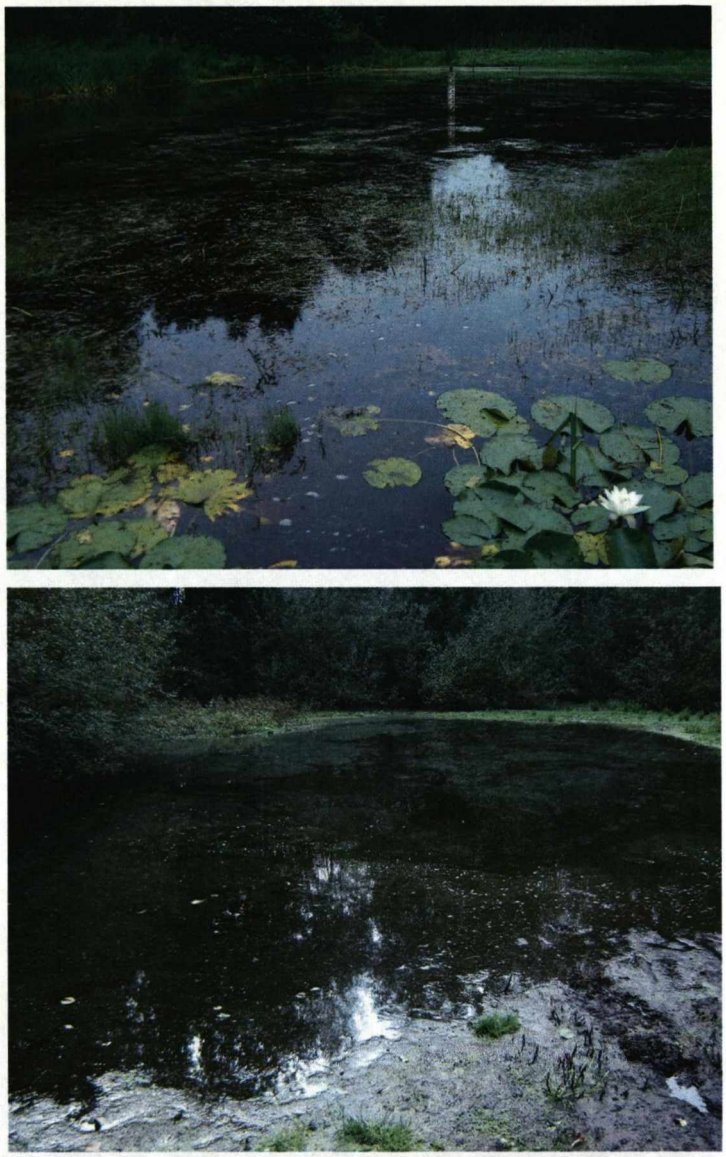


Figure 8.1 Comparison between a clear water state of pool 3 dominated by macrophytes (above) and a turbid phase of pool 7 with high densities of phytoplankton (below)

Water depths in all pools changed seasonally and simultaneously with increased levels in winter and decreases in summer. Pools in the area have become smaller and isolated and most pools dried out in summer, 2006. This suggested that the regional weather played a crucial role in the hydrological cycles of the pools. Fluctuation of water depth in temporary pools also showed strong impact on water chemistry. Low volume of water during the period before and after drying out resulted in increased nutrient concentrations.

Land use appeared to cause differences in water quality of the pools. In particular, pools located near the margin of Brown Moss Reserve had higher nutrient concentrations than those situated inside the nature reserve and this suggested that the impact from surrounding land was significant.

Other local factors also have a strong impact on the limnology of the pools. Pools with abundance of macrophytes such as pool 3 had clearer water and lower chlorophyll a concentration whereas pools 7 and 14 lacking aquatic plants had higher concentrations of chlorophyll a (see Chapter 2). Presence of macrophytes may have helped maintain a clear water state and in contrast, absence of macrophytes may have allowed phytoplankton to dominate and result in increase turbidity. Relatively more species and numbers of macroinvertebrates were also found in pools 3 and 6 in the presence of abundant plants since plants act as their important food sources and refuges. In contrast, it may have been a lower nutrient loading in the clear pools that limited growth of phytoplankton and permitted clear enough water to allow macrophyte growth, or a combination of both processes. Even in the least nutrient-rich pools, concentrations of total nutrients were comparatively high with TP values greater than 100 $\mu g\ L^{-1}$ and TN values greater than 1,500 $\mu g\ L^{-1}$ (Florida Lakewatch, 2000), however.

Zooplankters such as *Daphnia* were abundant also in pools 3 and 6 where plants were present and fish were absent. The presence of such large phytoplankton feeders was beneficial to pools because they reduced the risk of domination of the open water by phytoplankton especially in summer. In contrast, pools without large cladocerans such as pool 14 had high concentrations of chlorophyll a. Furthermore, fish have a strong effect on aquatic communities and structure food webs in pool 14. As predators, fish control populations of macroinvertebrates and large zooplankton and this may explain why macroinvertebrates and large phytoplankton were not abundant in pool 14. In contrast smaller zooplankton such as rotifers, copepods and *Bosmina* were present in pool 14 in large numbers. Absence of fish also appeared to promote other alternative predators such as *Chaoborus* to become important, as seen in pools 3 and 6. Pools that were dominated by macrophytes with the absence of zooplanktivorous fish appeared to have higher ecological values, in the sense of more abundance and greater biodiversity than that dominated by phytoplankton and with the presence of fish.

Although absolute dates could not be determined, paleolimnological studies revealed how pools at Brown Moss have changed through time. Also, this study was aimed to find out whether there have been changes in nutrient and biological status of pools. The results showed that nutrients and productivity in the pools have been increasing. The increase in nutrients and biological production was most probably due to the impact of land use around the site (Chapter 3). Dominant species of diatoms were good indicators of environmental conditions of the pools. *Frustulia rhomboids,* for example, was the main species of diatom in the bog pool (9) indicating acid conditions whereas *Melosira,* the most abundant species in sediments, indicated the nutrient-enriched condition of pool 6.

8.2 Important current issues at Brown Moss and implications for conservation and management

The maintenance a suitable environment is required by the conservation agencies at Brown Moss to ensure that decline in ecological value and problems of reduced biodiversity can be prevented. Proper management can minimize problems and maintain high ecological values and importance of the site. However, there are several issues of concern. In this section, the main current issues of water bodies at Brown Moss are revealed as well as implications for their conservation and management.

8.2.1 Nutrient input

Pools at Brown Moss are facing problems of high nutrients. The effective way to improve water quality in pools at Brown Moss is to reduce nutrient loading. As indicated by this study, the main sources of nitrogen of pool 6 came from shallow ground water and agricultural surface run off whereas internal sediments are regarded as a crucial source of phosphorus. Ultimate sources included birds and rain water.

Brown Moss is within a Nitrate Vulnerable Zone, as determined by the Environment Agency. Nitrate Vulnerable Zones are declared under the EU Nitrate Directive and land owners are required to restrict use of nitrogen fertilisers. The target concentration for groundwater is 7.5 mg N L^{-1} but this is far too high to be beneficial for conservation purposes (James et al. 2005; Moss, in press; Barker et al. in press). To

reduce N input from surface runoff and groundwater to suitable levels, immediate wetland buffer zones are required as natural filters around the edge of the site and pools. Such buffer zones have been effective elsewhere for nitrate removal (Lawrence et al. 1989; Blackwell et al. 1999; Young and Briggs, 2005). Terrestrial plants and trees in buffer areas can also take up nutrients. Nitrogen can also be reduced through denitrification while being transported.

Application of reduced quantities of fertilizers in agricultural areas may also help reduce nutrients lost and volatized, especially after heavy rain. Exposure of manure can allow ammonia from intensive livestock units to volatize. This source can also be reduced by storing manure in covered dumps and by adding water to it in store (Moss et al. 1997). Good agricultural practice is also important and can minimize the impact of nutrients. Examples of such good practice include ploughing across slopes or along contours, applying fertilizer in damp but not excessively rainy conditions and strip cropping of alternate bands of hay and green cover crops to avoid bare land in autumn. These methods are the most effective measures in reducing nitrate leaching (Grimble, 1994; Moss et al. 1997; Ministry of Agriculture, Fisheries and Food,1999). In practice, however, a great deal of nitrogen still runs off (Moss, 2002).

For phosphorus, seepage of domestic waste water appeared to increase P concentrations in shallow ground water and pools. Reducing contaminated effluent from residential areas can be achieved by an increased level of awareness and a conscious change in chemical use in the home around the site (Patterson, 2004). Using low or non-phosphorus cleaners and detergents and regular checking of pipes, hoses, nozzles and the repair of leaks can reduce problems. In fact, detergents in European countries now have much lower phosphorus concentrations than they did in the 1960s. Alternatively, the connection of houses to mains sewerage with sewage treatment works that have phosphorus stripping installed can be done. Direction of the soakaways or land drainage away from a valued site can also help if mains connection is too expensive.

Birds were the major source of external phosphorus to the main pool 6 at Brown Moss. The control of nutrients introduced by birds is not easy because at RAMSAR and SSSI site, birds are protected. However, there is still an indirect way that can be

applied to reduce numbers of waterfowl. As Brown Moss is a public site, visitors often feed birds. Giving birds supplementary food can encourage and attract more birds of relatively low conservation value, such as gulls, to colonize and stay around a lake. Therefore, discouraging people from feeding waterfowl by informing them about the impact of birds on water quality in pools may help reduce such problems.

In addition, birds do not spend all the time in the water but in fact, they often gather around the edge of water body when resting (*Figure 8.2*). Excreta produced by birds on the shoreline can be transported by surface runoff or rain to the pool. Therefore, regular removal of dried faeces left on shoreline especially during winter when densities of birds were highest may help reduce impact of nutrient input to pools at Brown Moss. Excreta collected could be given to farmers in the area or to visitors who can use it as a natural fertilizer for their gardens. However, such collection would be very costly and would itself cause disturbance and damage to plant communities.

Figure 8.2 Gathering of a large flock of Canada geese on shoreline in pool 6 in December 2004

Reduction of external nutrient inputs from ground water and birds alone may not be enough to restore the polluted pools and to maintain good water quality without other means of nutrient control. Effective ways of keeping pools in a clear water state with macrophytes dominating are to control of both external and internal nutrient sources. Sediments are the main immediate source of phosphorus, at least to the main pool.

Removal of the entire sediment or taking off the phosphorus-enriched surface layers may be required to reduce phosphorus internal loading. Moss et al. (1997) explained that the sediments can be sucked from the bottom of a lake and then conveyed by pipeline out of the lake. Proper disposal of sediments after being pumped is also required to reduce further environmental problems. In addition, sediment sealing using physical techniques such as plastic sheeting or using chemical substances such as aluminum sulphate or iron sulphate may be alternatives. However, chemical substances left in the system after treatment are likely to be harmful to the environment and organisms. Also, both physical and chemical isolation are temporary and rather costly.

8.2.2 Leaves and sediment deposition and shallowness of pools

Most pools at Brown Moss are temporary and becoming shallower since a recent map of Brown Moss in 1997 shows that pools are smaller and separated into small water bodies compared with those shown in a map in 1841 (*Figure 7.1, Chapter 7*). Reduction of water levels may also be likely to cause hydrological isolation (Reynolds, 1979). These pools have suffered from dramatically reduced water levels and eutrophication. Main causes of this issue could be the result of natural succession due to deposition of eroded sediment and leaf litter. Accordingly, to maintain water in pools especially in summer and to maintain them as aquatic habitats, pools may have to be deepened. However, before management work is undertaken, pools should be surveyed to assess their ecological status and presence of rare species of aquatic plants and animals. Certain areas of pools should be deepened through small scale dredging/digging to create and maintain water for summer and allow the other parts of the pools to remain as they are. This is because deepening pools to make the whole pool permanent is likely to change the aquatic plants and macroinvertebrate assemblages, and may result in the loss of uncommon species (Collinson et al. 1995; Nicolet et al. 2004).

Alternatively, one solution might be to excavate the site to create one or two bigger lakes as formerly. These would be large enough to allow aquatic vegetation over most of them and to have thick woodland right down to their edges. The disturbance of excavation would be considerable but the sites would recover.

Deepening of the pool may not be successful if plant materials and eroded soil still build up. Therefore, cutting down of some heavily overhanging and marginal trees around the pools may help reduce the rate at which ponds fill in. However, managing to cut too many trees back can disturb soil and allows more soils and nutrients to leak in. Falling leaves from trees around a pool do not seem to add significant amounts nutrients as trees take back most nutrients from their leaves before shedding them, so it is mostly their refractory carbon compounds that will fall into ponds (Williams et al. 1999). There are also other advantages of removal of overhanging and shading trees. Firstly, when shading trees are removed, light can penetrate to the water column and aquatic plants can start to develop and this will promote more species and numbers of plants in pools. Also, as a result of leaf deposition, some aquatic plants can hardly grow on the pond bottom and disappear from pools. Changing the substrate of pools from leaf litter to a silty and muddy bottom may help bring back freshwater rooted plants and increase the overall conservation values.

To reduce the amount of eroded soil going into the pools, buffer zones of semi-natural vegetation might be used again. Moss et al. (1997) thought complete coverage with vegetation of the soil in a buffer zone is desirable and vegetation that is actively building up biomass seems to be most effective at removing sediment. Widths of only 2m appear to be effective in retaining eroded particles in some places. Therefore, natural and existing vegetation such as shrub and small marginal plants around margin of pools should be kept as buffer zones.

8.2.3 Damage to aquatic plants

The experiment reported in Chapter 6 clearly showed that waterfowl had a strong impact on aquatic plant communities and biodiversity in pool 6. To reduce the effect of birds and protect aquatic plants from being damaged and lost, netting cages should be temporarily installed in the areas where important and rare plants such as *Luronium natans* occur, especially in winter when most birds gather in the pool as this has proven in this study to be effective. In summer when numbers of birds decrease, netting cages can be removed. However, a careful method is required to avoid disturbance of the site. Alternatively, zoning can also be used to permanently restrict access of waterfowl to some areas of the pool where rare plants are present. Lastly,

reducing numbers of birds by not giving them food may also help reduce impact of water birds on plants.

If plants are severely damaged or absent as a result of birds, re-establishment of the same species can be achieved. Such plants could be brought from the nearby areas or pools where they are abundant.

8.2.4 Invasive species

There are at least two main aquatic alien plant species at Brown Moss. These species include New Zealand pigmy weed (*Crassula helmsii*) present in pool 6 and water fern (*Azolla filiculoides)* in pool 2. Rapid and wide spreading of *Crassula helmsii* can reduce biodiversity of native plants whereas dense floating mats of *Azolla filiculoides* covering the surface can block out light to submerged plants and prevent exchange of gases with the air so that the underlying water can become de-oxygenated and noxious to aquatic animals (The Pond Conservation Trust, 2008). Therefore, stopping these species from spreading is needed to reduce further problems and impact on other pools at Brown Moss.

Williams et al. (1999) and Centre for Ecology and Hydrology (2008) suggest that chemical control of *Crassula helmsii* seems to be the most efficient way as mechanical control by cutting and tearing can spread the infestation downstream or re-infest the treated area. When *Crassula helmsii* is submerged, treating it with dichlobenil as Midstream GSR is recommended. Material on the banks or emergent forms should be treated with a formulation of glyphosate. Covering with black plastic or carpet could also be applied but seems to work only in small patches. Physical and chemical control of *Crassula helmsii* at the site has been carried out by Shropshire City Council, but it seems that further and regular treatment of *Crassula helmsii* is still needed.

Dense and spreading *Azolla filiculoides* indicates high nutrient concentrations in pool 3. Thus, minimising the amounts of nutrients in the pool using procedures described in 8.2.1 would be the best option to tackle the problem. Regular removal of floating plants can also be done by scraping them from the surface using a board or a sieve.

8.2.5 Visitors and local residents in the catchment

Anthropogenic impact is the main cause of environmental problems. Usually it is indirect through changes in nutrient loading or accidental introduction of alien species. Sometimes it is direct and deliberate. Lack of awareness or negligence of people can cause severe deterioration of an ecosystem. Each year there are many visitors coming to Brown Moss nature reserve and some visitors may cause environmental problems on purpose or by accident. For example, in 2006 while sampling at Brown Moss, I found gravels and other artificial materials from a fish tank dumped near pool 3. It was likely that somebody might have released unwanted fish or alien species to the pool and if this was the case, it could cause catastrophe to the aquatic system. Release of unwanted species of plants and animals in the wild seems to be the main cause of widespread distribution of alien species. Invasive organisms are very destructive to local species and can cause a serious threat to biodiversity as they have the ability to rapidly establish themselves and take over the new environments.

To avoid future problems of dumping of waste or illegal release of unwanted animals/plants to the site, more information about visitors' responsibilities could be displayed on site. If people visiting the site spot something unusual or see someone acting suspiciously, they should contact and report to Shropshire County Council straightaway. However, human nature does not seem to work in this way as they tend to avoid trouble and look away in the interest of self-preservation. There are many methods of visitor management such as using moats and ditches, which were successful in the Broads, England to keep visitors out of sensitive areas without them realizing that they are being managed.

Human activities in the catchment around the site are also likely to have an impact on pools at Brown Moss. Farmers from the agricultural areas within the catchment who use, spread or dispose of any substances that could pollute water, air or soil should be aware of their responsibilities and know about the causes and results of pollution. Accordingly, public participation and involvement of residents in the catchment are necessary to help protect Brown Moss and pools. In fact, Shropshire County Council has encouraged and supported the involvement of communities and individuals in

managing and conserving their local countryside and greenspaces (Edwards, 2007). An Open Day and guided walks are also regularly held to increase awareness of local people and visitors. A local interest group, Brown Moss Nature Reserve Users Association (BMNRUA) which is active in helping to manage the site, has also been set up.

Providing further information and advice on the appropriate management of the watershed (the Code of Good Agricultural Practice and reduction of point-source phosphorus as mentioned in 8.2.1) to local people living within the area may minimize the impact of nutrient-rich water that drains into the Brown Moss basin. Good agricultural practice describes the main risks of causing pollution from different agricultural and horticultural sources and means a practice that minimizes the risk of causing pollution while protecting natural resources and allowing economic agriculture to continue (Department for Food, Environment and Rural Affairs, 2008). Further information about the Code of Good Agricultural Practice can be viewed at www.defra.gov.uk.

8.2.6 Impact of poor water quality at Brown Moss on important species for conservation

Brown Moss is one of an internationally-important series of wetland sites that supports a number of rare species of plants associated with wetlands and an assemblage of aquatic invertebrates. It is a Special Area of Conservation under the EU Habitats Directive designated due to the presence of *Luronium natans*, which was first recorded at Brown Moss in 1955 by Edward Rutter (Lockton and Whild, 2003) and since then has been occasionally found in some years. *Luronium natans* was once a reasonably common species of the Shropshire meres and some rivers, but it had disappeared from most sites by the 1950s (Lockton and Whild, 2003).

Changes in trophic status of aquatic habitats from low nutrient concentrations to high levels of nutrients seem to be mainly related to disappearance of such species. Ecologically, *Luronium natans* occurs in habitats that are weakly productive, oligotrophic to mesotrophic, slightly acidic to circumneutral, shallow and with low alkalinity and mostly sandy sediment (Den Hartog, 1981; Roelofs, 1983, 1996; Pott,

1995). But now most pools at Brown Moss are hypereutrophic and this condition may not be suitable for this plant that grows in particularly low nutrient levels. Wide decline of *L. natans* has also been observed in other countries in Europe such as Germany and the Netherlands and it seems inevitably linked to the disappearance of suitable habitat type (Schaminée et al. 1992; Dierssen and Kaplan, 1993; UK Biodiversity Group, 1995; Greulich et al. 2000).

Several pools at Brown Moss are also important habitats and breeding sites for pond breeding animals. Great crested newt (*Triturus cristatus*), in particular, are protected under European and UK legislation and is one of species in the Shropshire Biodiversity Action Plan (Edwards, 2007). Pools with macrophyte dominance appear to be favourable for this species to breed because their eggs are laid on submerged aquatic plants. After hatching, they live in a pond as aquatic predators and therefore aquatic habitats dominated by vegetation are regarded as an important food source and good refuge for them. However, their populations have declined and this is again related to negative changes and loss of habitats such as pollution of breeding ponds and eutrophication (Krone, 2001; Kinne, 2006). Some studied pools such as 7 and 10 at Brown Moss are phytoplankton dominated due to excessive nutrient levels. These unfavourable habitat types lack submerged plant communities that provide food and refuges for such invertebrate species against predation, which could have a strong impact on overall populations of great crested newts in the area (*Figure 8.3*).

Figure 8.3 Unfavourable conditions of pool 7

The decline of important and rare species worldwide is associated with polluted habitats. Consequently, restoration of poor aquatic habitats at Brown Moss is essential and urgent to establish the return of *Luronium natans* and to maintain a long-term existent population of great crested newts. Restoration of freshwater habitats is discussed in 8.2.7.

8.2.7 Restoration and biomanipulation of hypereutrophic pools

Some pools at Brown Moss such as pool 7 and the one outside Brown Moss, pool 14, are phytoplankton dominated and turbid. Restoration of these pools is needed to switch them from turbid water to a clear water state with plant domination. Physical, chemical and biological techniques can be applied to improve quality of the aquatic ecosystems and steps of restoration are as follows.

Firstly, control of external and internal nutrient sources is required because reduction of nutrients, especially nitrogen and phosphorus can result reduce the development of phytoplankton. External nutrient control and reduction of internal loading by using buffer zones and sediment removal should be applied. Details of such controls can be viewed in section 2.1. After nutrient input is reduced, biomanipulation can be applied to help restore the system, but at this site, this is only relevant to pool 14 as other pools do not have fish communities already.

Biomanipulation is one of key procedures for lake improvement. It involves biological communities in the aquatic ecosystem and through using manipulation of the food web can effectively help improve water quality. Lakes with zooplanktivorous fish such as small roach, *Rutilus rutilus* (Linnaeus) or bream, *Abramis brama* (Linnaeus) have normally low densities of zooplankton grazers such as *Daphnia* because they are eaten by fish. As a result, without zooplankton, phytoplankton can develop and dominate water bodies. Therefore, to increase populations of phytoplankton feeders, zooplanktivorous fish have to be removed or piscivorous fish such as pike (*Esox lucius* (Linnaeus)) have to be added. Zander (*Stizostedion lucioperca* (Linnaeus)) is also used on mainland Europe where it is native.

Aquatic macrophytes in lakes are also involved in biomanipulation as they can control phytoplankton by taking up nutrients before these nutrients are used by algae or floating plants such as duck weed (Williams et al. 1999). In pool 7, some macrophyte species such as *Carex pseudocyprus, Oenanthe* aquatica, *Potamogeton natans and Ranunculus flammula* (Linnaeus) used to be present in 2003. However, recent eutrophication in the pool has resulted in the absence of those species. Therefore, re-establishment of vegetation in pool 7 after reduction of nutrient input is necessary. This is because macrophytes will help to restore the aquatic system and improve quality of water. The same species of those used to occur in pool 7 can be found and reintroduced to the pool from nearby water bodies in the area.

After plants have been re-established, protection of such plants is another important step because plants at this stage are vulnerable and susceptible to damage. To protect plants from herbivorous animals such as birds and reduce effects of waves, fences, cages or barrier could be installed. Once plants are established and well-rooted, protection can be removed. As aquatic macrophytes start to colonise, they can suppress phytoplankton. With the presence of macrophytes, populations of phytoplankton feeders such as *Daphnia* can start to develop.

Restoration may take time but once the restored system is stable, the mechanisms implemented can cause a switch from a turbid pool dominated by phytoplankton to a clear water and macrophyte-dominated state. Successful restoration will increase the conservation and ecological values of pools.

8.3 Future challenges for SSSIs and environmental conservation

As mentioned in Chapter 1, Sites of Special Scientific Interest (SSSI) are designed to protect important natural areas in the United Kingdom under nature conservation legislation. Brown Moss is one of them and is designated because of its scientifically important wetland habitat for plant and wildlife communities such as *Luronium natans* and *Triturus cristatus*. However, Brown Moss and many other freshwater nature reserves are experiencing declines in environmental quality. There are two main major future threats to SSSIs that need to be addressed and the ultimate future

challenges for SSSIs are to cope with such problems to maintain their ecological importance and good environmental quality for future generations.

The first future challenge of Brown Moss is to cope with pollution and nutrient problems caused by rapid population growth. The rising global population can create many problems such as rapid urbanisation, the demand for food production and resource scarcity. Increased food supply to support world population means free lands in catchment areas will be turned to agricultural fields and farms. Moreover, more fertilizers will be used to increase agricultural production as well as herbicide and insecticide that will be applied to cope with crop diseases. As a result of increased intensive agriculture, there has been concern over the effect of nutrients and pollutants on the environment from a watershed as whatever happens in the watershed can have a strong effect on SSSIs. A study of Willcock et al. (1999) showed that the measured specific yield of TN from the Toenepi catchment, New Zealand was the highest recorded in New Zealand studies and this reflects the high density of cattle in the catchment leading to high nitrate leaching losses. They also found that seasonal phosphorus peak concentrations in the stream coincided with times when fertiliser was being applied. In an agricultural catchment in southern Sweden, it was also reported that there was an overall correlation between amounts of pesticides used in the catchment and occurrence in the water samples and peak concentrations occurred during the spraying seasons and following runoff events (Kreuger, 1998). Our study has also shown that water quality of some pools at Brown Moss was as bad as those outside the nature reserve and this suggests that pools at Brown Moss may have been affected by nutrients from the catchment. For such a small catchment like Brown Moss with high slope, especially on the East of the site, the reserve area is likely to be negatively impacted by activities within its catchment.

In the future, if more quantities of nutrients such as nitrogen and phosphorus are still used to increase crop production, they are likely to cause more severe problems to the aquatic ecosystem. Recent studies have shown that high concentrations of nitrate approximately about 1 mg $NO_3N\ L^{-1}$ increased the rate of loss of aquatic plant species from freshwater communities (James et al. 2005; Barker et al. in press). Another crucial issue related to increased nutrient concentrations is eutrophication which appears to have had a cascading effect through the whole system. Contamination of

herbicides and pesticides from catchment areas in runoff or ground water entering water bodies can also reduce biodiversity. Several studies clearly showed that increased pesticides have a strong impact on the decline of aquatic animal populations. For example, Relyea (2005) found that carbaryl and malathion, globally common pesticides normally used in garden and crop land, affected the zooplankton communities by eliminating cladocerans and at high concentrations (1 mg L^{-1}); carbaryl can completely wipe out nearly all species of zooplankton. In addition, carbaryl was also lethal to gray tree frog tadpoles (*Hyla versicolor* (LeConte)) (Relyea and Mills, 2001).

The management of the site and pools within a nature reserve alone will not able to cope with the real sources of problems. In the future, whole catchments should be protected under the law and legislation to ensure that activities within the watershed will not cause any environmental problems to the sites. In addition, inadequate or lack of scientific monitoring such as water quality of SSSIs can result in rapid deterioration of pools. Polluted freshwater habitats can lead to loss of biodiversity of aquatic plants and animals and this can significantly decrease the ecological value of the site. In fact, Natural England has a statutory duty to notify SSSIs and to ensure that England's natural environment is protected and conserved. However it seems that Natural England does not meet its goals in the aspect of limnology since a survey of freshwater SSSIs in 1995 showed that up to three-quarters were affected by eutrophication (Carvalho et al. 1995) which was consistent with this study revealing that most pools at Brown Moss were hypereutrophic.

The limitations on this appear to be complex and may include shortage of funds, lack of expertise and conflicts between use of the site for conservation and recreation. Large bird populations and diverse plant communities may, for example, be incompatible. However, transferring more assistance to local organisations responsible for SSSIs from national authorities such as the government and Natural England in term of scientific and financial support may help improve environmental conditions of SSSIs as local authorities are closer to the site and problems. Moreover, more in-depth research about freshwaters and regular water quality monitoring such as concentrations of nitrogen, phosphorus and chlorophyll a in all pools should be implemented at aquatic SSSIs. With the help of all parties, conservation goals can be

achieved and this could improve quality of freshwater habitats and bring back *Luronium natans* to Brown Moss and help populations *Triturus cristatus* to flourish even more.

Internationally, there is also an attempt to protect water management through river basin planning. The Water Framework Directive, for example, is a recent piece of European legislation which promotes a new holistic approach to improve and protect inland and coastal waters, especially from diffuse pollution in urban and rural areas, through better land management (Environment Agency, 2008). This new approach also promotes wiser, sustainable use of water as a natural resource and better habitats for wildlife that live in and around water. In England and Wales, the Environment Agency is the sole competent authority for implementing the Water Framework Directive and its responsibilities include analysing the characteristics of the 11 River Basin Districts in England and Wales, assessing the impact of human activity on the water bodies within these districts and taking the lead in drawing up the Programme of Measures and co-ordinating the actions within it. The current Water Framework Directive offers a basis for this, but unfortunately small water bodies such as those at Brown Moss can be excluded from its provisions. In any case the Directive may not prove to be as effective as intended (Moss, 2008). More information about the Water Directive Framework can be viewed at www.euwfd.com.

Another important challenge for SSSIs in the future is climate change. Climate is a global concern and changes could have catastrophic and unprecedented impacts on all aspects of life on earth. In freshwaters, rising temperature appears to cause direct impacts as the hydrological cycle is driven by the climate. Rising temperature could perhaps make a period of drying out in temporary standing waters last longer and this will affect life cycles of aquatic plants and animals. Drought can also occur more often and this can lead to the disappearance of lakes as a result of increased evaporation relative to precipitation (Magnuson et al. 1997). Disappearance or extinction of aquatic plants and animals that are susceptible to such severe conditions would also be expected. Also, increased temperature could rapidly warm up surface water and thus reduce oxygen concentrations in deeper hypolimnetic layers that could kill aquatic flora and fauna (Magnuson et al. 1997; Mulholland et al. 1997). Depletion of oxygen in deeper areas could also be the result of an increased rate of bacterial

activity in the hypolimnion waters and sediment due to warmer lake temperatures (Blumberg and DiToro, 1990). Nutrient concentrations and pollutants would increase as lake volumes shrink while the size of drainage basin and inputs remains unchanged and these abiotic changes would mean a general shift towards species characteristics of eutrophic and contaminated lakes (Magnuson et al. 1997).

Warmer water in summer could also promote blooming of potentially toxic blue green algae (cyanobacteria) that are harmful to plants and animals. Johnk et al. (2008) predicted by using a model, climate change is likely to yield an increased threat of harmful cyanobacteria such as *Microcystis* in eutrophic freshwater ecosystems due to high temperatures that favour cyanobacteria directly, through increased growth rates. Moreover, high temperatures also increase the stability of the water column, thereby reducing vertical turbulent mixing, which shifts the competitive balance in favour of buoyant cyanobacteria. Through these direct and indirect temperature effects, in combination with reduced wind speed and reduced cloudiness, summer heatwaves boost the development of harmful cyanobacterial blooms. Briand et al. (2004) found that the colonization of mid-latitudes by *Cylindrospermopsis raciborskii* (Wolosz) (a tropical bloom-forming cyanobacterium recently identified in several temperate areas) may result from the global warming, which provides this species with better environmental conditions for its growth.

Good management and planning can minimise such impacts. Regular and appropriate water quality monitoring programmes of aquatic habitats should be implemented to assess change or deterioration. The data will serve as a basis for effective and fast decision-making and can be used to design and implement suitable solutions to problems. Reduction of anthropogenic impact can also help reduce impacts of climate changes on aquatic ecosystems. These strategies include restoration of riparian vegetation around the site and maintaining aquatic habitats in good conditions.

In conclusion, it will be a challenge for the future to conserve natural habitats and maintain them in good conditions while world populations and environmental problems are rising. Successful management in the future needs cooperation of all parties from policy makers to visitors using the site and local residents living within the catchment. More concrete actions in freshwater protection and improvement, from

the government and appropriate environmental agencies will also be needed. In addition, environmental responsibilities and awareness of people are key driving factors for environment protection. Regular scientific investigation of water quality such as nutrient status and in-depth research is also important in the future and should play more roles in local management and environmental conservation. Protection and maintaining the interest and favourable status of the internationally significant wetland for which Brown Moss was designated is the ultimate goal of conservation.

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